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- *Synergies and Conflicts between Noise and Air Quality Action Plans*
- *Verification of an Atmospheric Emissions Inventory*
- *Factors affecting Inter-annual Variability of NO_x and NO_2 Concentrations*
- *NO_x Chemistry on Power Station Plumes*
- *Impact of Air Quality Standards on Vegetation and Ecosystems*

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CONTENTS

The National Society for Clean Air and Environmental Protection produces information, organises conferences and training events, and campaigns on air pollution, noise and environmental protection issues. Founded in 1899, the Society's work on smoke control led to the Clean Air Acts. More recently NSCA has been influential in developing thinking on integrated pollution control, noise legislation, and air quality management.

NSCA's membership is largely made up of organisations with a direct involvement in environmental protection: industry, local authorities, universities and colleges, professional institutions, environmental consultancies and regulatory agencies. Individual membership is also available to environmental specialists within industry, local authorities, central government, technical, academic and institutional bodies.

Members benefit from joining a unique network of individuals who share an interest in a realistic approach to environmental protection policy; from access to up-to-date and relevant information; from reduced fees at NSCA conferences and training events. They contribute to NSCA's regional and national activities; to environmental policy development; to translating policy into practice; to the Society's wide-ranging educational programmes.

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Editorial

A BOOK OF ALL THINGS

Architects have a book called The New Metric Handbook. It provides measurements for such things as amount of legroom needed in a cinema, the turning space needed in a car park, the size of a standard WC. It is essentially a data source book allowing you to design, for example, an office building safe in the knowledge that everyone will have sufficient work space and getting out of the car park will be easy (in theory, at least).

The collation of standard, common data is not, therefore, new, nor is it rocket science. So it is a source of some frustration that, in an age when information on pretty much everything is apparently available at the touch of a search engine, there are reports that local authorities have "lost" their data from the first round of air quality review and assessment and that the data needed for noise maps (traffic flows, etc.) is proving "difficult" to come by. Public registers for IPC, IPPC and LAAPC are generally in a state somewhere between appalling and wretched. Despite virtually all "stage 3" review and assessment studies being carried out using computers, very few have been submitted in electronic form. If you want access to this data, you had better be prepared to put in the hours, days and weeks.

Not that the blame for this state of affairs should rest with local authorities. They are being asked to carry out an ever increasing number of studies, assessments and surveys of their areas on environmental and other issues. Little thought appears to be given to where the base data will come from and what will happen to it when the next policy initiative arrives. A lack of coordination means that wheels are constantly reinvented, albeit with a different number of spokes, to suit the needs of different surveys and assessments. A "not invented here" culture pervades and the software is never compatible.

Instead of all this reinvention, wouldn't it be helpful if local authorities were encouraged, even funded, to produce a New Metric Handbook for their own areas, a book of all things to which they could turn for information on traffic flows, population distribution, building types, energy use, and so on. Local authorities are, among many other things, massive repositories of data on a huge variety of topics. In fact, it is hard to think of any local information which they do not hold, although it's much harder to actually find it and turn it into something useful. If this information could be collated, it would form a powerful tool, not only for the preparation of noise maps, air quality assessments and the like, but also for research into the state of local environments.

Wishful thinking, perhaps, but greater planning, coordination and communication between different policy areas could potentially pay very large dividends in terms of time saved by those who have to implement such policies, i.e. local authorities.

In this issue, we carry a number of articles critically concerned with data. Greg Archer and colleagues report on the synergies between air quality and noise, showing that in many cases the sources for both (and therefore much of the usage information) are common. Clearly, good quality data is critical to the assembly of reliable atmospheric emission inventories, and Matt Ireland's article on the verification work carried out in Kent shows the need to have this in an accessible and replicable form. The last article of this issue, by Nigel Bell, illustrates a subject where two different environmental issues, air quality and biodiversity, overlap and where a "book of all things" might be useful. The third and fourth articles, by David Carruthers and Tim Hill, revisit NO_x and NO_2 chemistry, this time from point sources, another area where more data is needed.

REPORTS

Synergies and Conflicts Between Noise and Air Quality Action Plans

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Abstract

The similarities between important sources of air pollution and noise mean that there are synergies or conflicts between measures designed to improve local environmental quality. This paper examines these interactions and concludes that the overwhelming majority of potential mitigation measures are unlikely to cause conflicts and, for some, notable synergies exist. Case studies demonstrate that local circumstances significantly affect the outcomes of any mitigation measures and scheme-specific assessments are therefore essential to optimise the design of the measure and ensure it achieves its intended outcomes.

Overview

Local air quality management and the provisions of the EU Directive on Environmental Noise require mapping of local air pollution and noise, and development of action plans where necessary. This paper presents the result of a study to investigate potential synergies and conflicts between noise and air quality action plans and the extent to which these impacts are sensitive to local circumstances. The study was undertaken by WS Atkins Environment for the DETR (now DEFRA) and the Devolved Administrations and is available from the DEFRA website.[1]

There are similarities between many of the sources and potential mitigation strategies to address air pollution and noise. Important sources of both noise and air pollution include: road traffic, railways, airports and industry including quarries and landfill sites. There are a wide range of local mitigation measures that can be used to reduce the air pollution and noise from traffic and industrial sources, many of which can be effectively implemented by local authorities. Local measures are available to reduce air pollution and noise from airports but these are mostly within the remit of the infrastructure or transport operator. Railways are an important noise source but in most circumstances do not make a significant impact upon air quality. There are relatively few measures to address local railway noise but noise barriers, where these can be installed, can be highly effective.

The overwhelming majority of potential mitigation measures are unlikely to cause conflicts between the objectives of reducing air pollution and noise. Some measures, particularly those concerned with managing levels of activity, such as reducing local traffic flows, benefit air quality and noise – although a reduction of at least 25% in traffic is required to achieve measurable benefits for noise. Other potential mitigation measures that can reduce both air pollution and noise are restrictions on heavy vehicles and reducing speeds on motorways and dual carriageways. Strategies to increase the separation between the source and sensitive receptors, such as by the use of "buffer" zones or traffic bypasses can also benefit both air pollution and noise.

There are some potential mitigation measures in which the objectives of reducing air pollution and noise conflict. Most notably, measures to reduce average speeds of traffic in urban areas (where average traffic speeds are lower), whilst generally benefiting noise, will increase emissions of air pollution. Also, zoning of industry, whilst enabling noise sources to be located away from sensitive receptors, will concentrate sources of air pollution and may cause a worsening in air quality downwind of the site.

Traffic Management Case Studies

Illustrative case-studies undertaken as part of the study have shown that for many measures, local circumstances will significantly affect the impacts and effectiveness of the proposed measures. It is therefore essential that local assessments are undertaken in order to optimise the design of the scheme, ensure it achieves its intended outcomes and that the costs are justified. For each case study impacts upon air quality were assessed in terms of changes in emissions or concentrations of PM_{10} and NO_x using emission rates and assessment approaches taken from the Design Manual for Roads and Bridges[2] with calculations performed for 2005. Noise calculations were conducted to determine L_{eq} or L_{10} (the level exceeded for 10% of the time) using the prediction methods contained in the Department of Transport's memoranda Calculation of Road Traffic Noise[3] (CRTN). CRTN uses a 2-category definition for the traffic composition – cars/light vans and heavy duty vehicles (HDVs). Where a mitigation strategy related to individual classes of vehicle within the HGV category, such as buses or different weights of lorry, the noise source data contained in The Noise Advisory Council publication 'A Guide to

Measurement and Prediction of the Equivalent Continuous Sound Level L_{eq}'[4] was used.

The effectiveness, impacts and costs of most mitigation measures are very sensitive to local circumstances and the manner in which the measures are implemented, will depend upon local circumstances. The results presented in this paper are therefore illustrative; a robust assessment of the local environmental quality, the principal sources affecting these areas and all the impacts, costs and benefits of the proposed mitigation measures are therefore an essential pre-requisite of any effective action plan.

Case Study 1
Impacts of Changes in Vehicle Flows and in % HGVs

For air quality, changes in emissions from vehicles on a road are directly proportional to changes in traffic flows (other parameters being constant). However, the proportionate changes in pollution concentrations will usually be significantly less than those in emissions due to the contribution from other "background" sources. For noise, significant reductions in traffic flows of at least 20% are needed to produce a 1dB(A) change in noise (with constant speed and percentage HGVs). Greater reductions in vehicle flows are therefore required to achieve noticeable differences in noise than for air pollution. Relatively small changes in the proportion of HGVs will produce relatively greater effects for noise than changes in vehicle flows.

Vehicle access restrictions can be an effective means of reducing traffic flows and potentially reducing air pollution and noise, and Traffic Regulation Orders (TRO) can be implemented for the purpose of improving air quality such as in a Low Emissions Zone. Table 1 shows the impact upon noise and vehicle emissions of restricting access to a range of vehicle types. The assessment assumes these vehicles are not replaced by other traffic and some of the estimated reductions will therefore be due to lower traffic flows.

Each of the restrictions produces sizeable reductions in emissions. The results are not significantly different for the two speeds assessed. Noise levels are also reduced as the larger lorries are removed from the traffic stream. For example, at an average speed of 50 km/hour, removing only the larger lorries would reduce noise levels by 0.9 dB(A), whereas banning all commercial vehicles other than buses and light vans would reduce levels by 2.3 dB(A). The assessment demonstrates lorry bans have the potential to produce worthwhile improvements in both air quality and noise, especially if used in conjunction with more stringent speed restrictions.

Case Study 2
Bus use

Changes in vehicle flows can also be achieved by a range of approaches including green commuter plans and provision of improved public transport to encourage a modal shift from private vehicles. Many of these are designed to use buses to transport former car occupants but the net impacts of a scheme are highly dependent upon local circumstances as buses tend to emit more air pollution (NO_x and PM_{10}) and be noisier than cars. In order for schemes that seek to encourage a modal shift for private cars to buses to be beneficial for noise and air pollution a minimum number of cars must therefore be removed from the road network for each additional bus. Figure 1 shows the minimum number of cars that must be removed from the road – per additional bus, in order to achieve improvements in noise and emissions of NO_x and PM_{10}. This assumes each bus holds 55 passengers and cannot carry more than the equivalent of 27 cars of two passengers. The calculations used to derive Figure 1 are highly sensitive to the assumptions employed and demonstrate the importance of local assessments to predict likely impacts.

Table 1 – Effect of Vehicle Access Restrictions on Emissions and Noise

	NO_x (% change in emissions)		PM_{10} (% change in emissions)		Noise (change in dB(A))	
Vehicle Ban	**30 km/hr**	**50 km/hr**	**30 km/hr**	**50 km/hr**	**30 km/hr**	**50 km/hr**
HGVs [1]	-58	-52	-38	-33	-4.3	-2.3
HGVs [2]	-	-	-	-	-3.7	-1.5
Articulated HGVs [3]	-24	-29	-15	-11	-2.4	-0.9
Pre-Euro I	-19	-25	-18	-19	-	-
Pre-Euro II	-32	-37	-38	-37	-	-
Pre-Euro III	-71	-68	-66	-70	-	-

[1] for noise ›1525 kg ulw, for air ›3.5 tgw, these are equivalent

[2] for noise › 2 axle ›3t ulw (broadly equivalent to 7.5tgw)

[3] for noise ›= 3 axle, for air – no articulated HGVs

Note: An air quality assessment could not be undertaken for HGVs ›7.5 t gw as no emission rate information is available for medium goods vehicles (3.5-7.5t gw). A noise assessment could not be undertaken for banning Pre-Euro I, II or III vehicles as no noise data are available.

Addition of buses could not be beneficial at bus speeds at or below 20 km/hour, when more than 27 cars would need to be removed to achieve a decrease in emissions. If the buses are not filled to capacity, higher speeds are needed for the scheme to be beneficial. In this example, if the bus is less than 44% full, it cannot be beneficial in terms of emissions at any speed. The limiting parameter at all speeds is PM_{10} and where cleaner buses (such as Euro III or Euro IV) were used a smaller number of cars would need to be removed for the scheme to be beneficial. Use of cleaner buses (or ideally electric) is therefore highly desirable where the intention is to improve local air quality by reducing vehicle flows.

For noise, at average vehicular speeds in excess of about 35 km/hour, the noise level/speed relationships are similar for both cars and buses. No further noise advantage is therefore to be gained by increasing average speeds above this value. The greatest benefits to noise are expected to occur at speeds of 35 km/hr (22 mph) or more where each additional bus would need to replace only four cars for there to be equivalent noise emissions.

Case Study 3
Use of Bus Priority Lanes

Figure 1 demonstrates improvements to air pollution and noise are likely to be greatest at speeds equal to or above 35 km/hour and use of bus priority lanes can be a highly

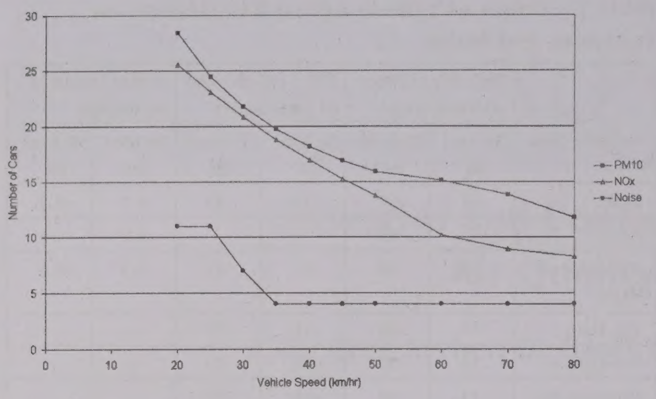

Figure 1 – Minimum Number of Cars to be Removed, per Additional Bus, to Achieve Improvements in Air Quality and Noise

effective means by which to increase bus speeds and service reliability to promote use. Introducing a bus lane on an existing road is also likely to lead to reducing the number of lanes available to other vehicles, increasing congestion. The resultant change in noise and air quality due to introduction of a bus priority lane will therefore depend upon relative vehicle and bus speeds and flows before and after the bus lane was introduced. The effects of five scenarios on emissions and noise are shown in Table 2.

If the bus priority lane does not result in vehicle speed changes, any change in noise and air quality is expected to be small due to the slightly increased distance between the bulk of the traffic and receptors (assuming that the bus lane is adjacent to the pavement). In the unlikely event that the bus lane was very heavily trafficked (by buses and/or taxis), emissions of PM_{10} and NO_x could potentially increase in the lane closest to kerb. All of the scenarios where there is a decrease in 'other vehicle' speed are predicted to lead to an increase in emissions that is greater than any decrease in bus emissions. Scenario 4 was predicted to reduce emissions due to the increase in bus speeds and consequent reduction in bus emissions whilst the other vehicle speed remained constant. The overall change in emissions will be dependent upon the bus proportion in the fleet, with higher bus proportions expected to give a more favourable result.

Sound power levels generally decrease with decreasing vehicular speed. However, below a certain speed (that varies with vehicle type) there is no further decrease. These critical speeds are about: 27-29 km/hour for cars and medium sized lorries; 39 km/hour for buses and 40-47 km/hour for the heavier lorries. Any increase in bus speed above the optimum would lead to increased noise from this source. Table 2 shows that the noise increase due to increases in bus speed will be offset to varying degrees by a reduction in the speed of the other vehicles. The result may vary between a small net noise increase and noise reductions of up to about 3dB(A).

Table 2 – Expected Changes in Emissions and Noise due to Bus Priority Lanes

Scenario Number	Scenario Speeds (km/hr) All before	Bus after	'Other' after	Emissions (change as %) NO_x	PM_{10}	Noise (change in dB(A))
1	25	50	20	+5	+3	+0.5
2	40	40	25	+19	+17	-2.6
3	40	45	30	+10	+8	-1.6
4	40	50	40	-2	-2	+0.2
5	40	60	25	+14	+15	-1.8

Case Study 4
Changes in Speed

Changes in speed, such as from the introduction of new speed limits, is another area of potential conflict between measures to improve noise and air pollution. The optimum speed for noise varies by fleet composition, but is in the range 20-40 km/hr. The optimum speed for NO_x is about 70 km/hr, whilst for PM_{10} emissions increase below 90 km/hr. Table 3 shows the expected changes in emissions and noise due to reduced vehicle speed for four scenarios (for a fleet composition with 10% heavy vehicles).

Each of the scenarios is expected to be beneficial to noise whilst only two, (involving a reduction from 70 mph), would be beneficial for air pollution. Speeds in urban areas are often less than 50 km/hr and some local authorities, generally for road safety reasons, are considering reducing the speed limit in residential areas from 50 km/hr to 30 km/hr. Reducing average vehicle speeds in this manner could reduce noise levels by 0.8 dB(A) but would increase emissions of some pollutants by over a quarter. This would need to be a consideration in any proposed changes to speed limits in urban areas. On motorways and rural dual carriageways, with traffic travelling in excess of 70km/hr, speed restrictions would be beneficial for both noise and air quality.

Conclusions

There are a wide range of local mitigation measures that can be used to reduce the air pollution and noise from traffic, airports and industrial sources. There are relatively fewer local measures that can be adopted to reduce noise from railways (air pollution is not usually a major issue from railways). Local authorities have a wide range of effective powers that can be used to address many local traffic and industrial sources of air pollution and noise but have less direct influence over railways and airports.

For the overwhelming majority of potential mitigation measures there are unlikely to be conflicts between measures to reduce air pollution and noise and for some measures notable synergies. However, local circumstances will significantly affect the outcomes of any new measures, and scheme-specific assessments are essential to optimise

Table 3 – Expected Changes in Emissions and Noise due to Reduced Speed Limits

Scenario	Emissions (change as %) NO_x	PM_{10}	Noise (change as dBL_{A10})
30 mph to 20 mph	+27	+27	-0.8
40 mph to 30 mph	+7	+9	-1.2
70 mph to 60 mph	-8	-4	-1.2
70 mph to 50 mph	-15	-4	-2.5

the design of the measure, ensure it achieves its intended outcomes and that the costs are justified.

Although the majority of traffic management measures designed to reduce local air pollution or noise do not have a detrimental impact upon the other criteria, in some circumstances conflicts may exist. In particular, measures to reduce average speeds of traffic in urban areas whilst benefiting noise will increase emissions of air pollution. Also, measures designed to reduce private vehicle use, such as park and ride, green commuter plans or bus lanes may improve environmental quality in some circumstances; in others, the impacts may be negative, particularly on air quality. Local circumstances will significantly affect the outcomes of any new measures, and scheme-specific assessments are essential to optimise the design of the measure, ensure it achieves its intended outcomes and that the costs are justified.

In most urban locations, implementation of a single mitigation measure is unlikely to achieve the required improvement in environmental quality for air pollution and/or noise. This is partly since a diverse range of sources affect environmental quality; but also since a single measure is unlikely to achieve the required level of improvement. This is particularly true in areas dominated by road traffic sources in which a package of complementary measures within an action plan is the preferred approach. Examples of combinations of measures likely to be most effective for traffic sources are:

- Bus Priority lanes / High Occupancy Vehicle Lanes integrated with a Park and Ride scheme utilising low emission vehicles.
- Construction of a bypass with speed restrictions, quiet road-surface and barriers (if there are properties nearby), together with controls on vehicles entering the bypassed area.
- Urban traffic control (UTC) systems that can be used to optimise traffic speeds and prioritise flows for public transport lanes.

In some locations, fiscal incentives such as congestion charging or work place parking charges may be required to encourage motorists to use public transport. The benefit of a suite of measures is likely to be greater than the sum of each individual measure.

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Verification of an Atmospheric Emissions Inventory

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This short paper describes some of the verification techniques used to evaluate the Kent and Medway Emissions Inventory. Results are presented showing the spatial and temporal extent of the inventory.

Introduction

An atmospheric emissions inventory has been compiled for Kent and Medway in the south east of England. The study area is diverse with significant historic and projected change. The presence of major transport corridors, industry, small and larger urban areas, and contrasting rural areas suggests the area is expected to illustrate most of the issues associated with air quality management.

The inventory includes annual mean estimates of emissions of key pollutants associated with local air quality (nitrogen oxides, NO_x, and particles, as PM_{10}) and climate change (carbon dioxide, CO_2 – the principal greenhouse gas) for scenario years between 1990 and 2020. The inventory includes emissions disaggregated to 250m by 250m grid squares and hence, is of a much finer resolution than the 1km by 1km national and urban inventories published for the UK.

The methodology for compiling the inventory is not detailed in this paper but follows the usual practice of coupling activity data with emission factors. The inventory was compiled from the 'bottom up', to allow direct linkage with land use, transportation and economic forecast data. Through the use of nationally available and published emission factors the resultant inventory of annual mean emissions of NO_x, PM_{10} and CO_2 is consistent with other UK national and urban emission inventories.

The objective of this paper is to describe some of the verification techniques used to evaluate the emissions inventory and present some of the findings.

Verification of Emissions

A review of the literature describing the compilation and verification of emission inventories has been undertaken. The techniques used for the Kent and Medway inventory are summarised in Table 1.

Comparison with other Inventories

Completeness and consistency was considered by comparing the method with those of the NAEI and urban inventories. The results of estimating NO_x, PM_{10} and CO_2 emissions from all sectors in Kent and Medway are summarised and compared to estimates derived from national and urban studies in Tables 2 and 3.

The study area and population for each study are included where available for comparison. The emissions are all reported in kilotonnes per annum with NO_x emissions generally being a factor of 6-12 greater than PM_{10} emissions and CO_2 emissions typically a factor of 200-300 greater than NO_x emissions.

The inventory of Kent and Medway therefore appears to be consistent when compared to other UK national and urban studies both in terms of total and sector contributions. Moreover, the most direct comparison, with equivalent NAEI data, suggests a high degree of consistency.

These summary data have been subdivided into source category (data for PM_{10} provided in Table 5 by way of example). Some source categories have been aggregated to allow comparison between inventories. All the inventories show some consistency in identifying road transport and industry as the dominant sources of NO_x and a generally similar pattern for PM_{10}. Industrial NO_x emissions in Kent are estimated to be in the order of 59-63% similar to Swansea and Port Talbot, reflecting the significant industrial element to land use in Kent. The proportion of PM_{10} emissions from industry in Kent (64-68%) is also comparable to Swansea and Port Talbot. Road traffic in Kent contributes approximately 34-38% of NO_x and 32-35% of PM_{10}. As may be expected, these proportions are somewhat lower than urban areas but are broadly comparable to UK data. Although both industry and traffic are significant contributors of CO_2, all the inventories reflect the importance of emissions from the residential and commercial sectors.

Spatial Analysis

The annual mean emissions of NO_x are generally less than two tonnes per 250m by 250m square in rural areas and between two and three tonnes per 250m by 250m square in urban areas. The road network is clearly highlighted. Maximum emissions per 250m by 250m square are associated with the local airports, suggesting further assessment is required although this source represents less than 1% of total emissions for Kent and Medway. The pattern of PM_{10} and CO_2 emissions is similar to that for NO_x although the urban areas are more distinguished for CO_2, reflecting the relative contribution made by building related emissions as a proportion of total CO_2 for Kent and Medway. Building related emissions of NO_x for Kent and Medway are estimated to be 0.8% compared to 15.6% of total CO_2 emissions.

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[3] Kent County Council

Uncertainty Analysis of Emissions

Following the Tier 2 method described by the IPCC (2000) in its Good Practice Guide (see also Salway *et al*, 2001) and the work of Sadler (1998) as a basis, estimates of tolerance (range of uncertainty) for each emissions source have been compiled in conjunction with other published data and judgement – see Table 6. These estimates of tolerance are applied to current emissions (1997-2000) assuming a minimum value of 0, and a further margin applied for extrapolation to other years (Table 7). Evidently, the tolerances provided could be debated but, with the absence of specific published data, are considered sufficient for demonstrating some consideration of uncertainty. These tolerances have been used as input for a Monte Carlo analysis to determine the overall degree of uncertainty of the emissions inventory. For each year, the total emission for each source category was calculated and a random value within the relevant tolerance generated. Following the method described by Salway *et al* (2001) some 10, 000 random values were generated for each source and the 97.5^{th} percentile confidence limit calculated to provide a range of uncertainty, i.e. the range within which 97.5% of the values lie (i.e. the mean ± twice the standard deviation). The estimated ranges of uncertainty, expressed as twice the standard deviation divided by the mean, for the emissions inventory are summarised in Table 8.

The results of this uncertainty analysis are comparable with national estimates for NO_x and PM_{10} (Goodwin *et al,* 1997) suggesting the inventory is sufficient for local to regional scale studies. Although the uncertainty of this inventory for CO_2 is comparable to NO_x and PM_{10}, it is considerably larger than the 5% reported for national emissions of CO_2 (Goodwin *et al*, 1997). This may be expected, as the national estimate is based on the 'top down' approach of estimating CO_2 emissions directly from fuel consumption which, being taxed, is relatively well defined. The methodology is sensitive to the assumed margins of tolerance applied (i.e. Tables 6 and 7) which, in the case of this work, and that of Goodwin *et al* (1997) and Salway *et al* (2001) is largely based on judgement. Experience in compiling this inventory suggests reducing the degree of uncertainty to less than reported nationally is unlikely without significant additional resources to compile more accurate emission factors and activity data.

This method does not provide a measure of spatial error although the discussion on spatial analysis above provides a qualitative assessment of spatial consistency, concluding that the pattern of emissions follows the underlying land use pattern. The difference in spatial resolution between this inventory and the equivalent data from the NAEI precludes a spatial comparison as undertaken by Lindley *et al* (2000).

A final means of verification was attempted by comparing the ratios of observed concentrations and estimated emissions at grid squares with concurrent data (after Funk *et al,* 2001). The results, summarised in Table 9, are inconclusive. The ratio of observed NO_x to PM_{10} concentrations are in the order of 2-5 whereas the ratio of NO_x to PM_{10} emissions range from 2 to 40. This is most likely due to the large background component of PM_{10} which typically represents 80-90% of background observations and 40-50% of kerbside observations.

Inventory Results

Trends in emissions of NO_x, PM_{10} and CO_2 by sector for the years 1990 to 2020 are illustrated in Figures 1 to 3. Emissions of NO_x (Figure 1) are projected to decline from 124.7 kilotonnes in 1990 to 74.16 kilotonnes in 2010 with only a very small further reduction by 2020. Although emissions from industry decline slightly, the most significant reduction is in road transport – from 52.8 kilotonnes in 1990 to 15.4 kilotonnes in 2020, a fall of 71% despite high growth in road traffic. Across all sectors, NO_x emissions in 2010 are expected to be 59% of emissions in 1990 and comparable to the emissions reduction target for reducing emissions between 1980 and 2010. The trend in PM_{10} emissions follows a similar pattern although significant reductions from industry, associated principally with fuel switching from coal and oil to gas, are observed between 1990 and 1997. Again, the most significant reduction is observed in road transport contributing to total emission reductions from 11.6 kilotonnes in 1990 to 5.3 kilotonnes in 2020 – a reduction of 54%. Stedman *et al* (2001) provide similar projections for the UK as a whole. The UK total urban road traffic NO_x and PM_{10} emissions in 2020, for example, are projected to be 19% of 1990 emissions, with UK total non-road NO_x and PM_{10} emissions 58% and 34% of 1990 emissions, respectively.

Emissions of CO_2 in Kent and Medway are predicted to fall from 9.3 megatonnes in 1990 to 9.0 megatonnes in 2010 and rise thereafter. The fall between 1990 and 2010 is 2.5% – well below the UK Government target of 20% and the national commitment to the Kyoto Protocol of 12.5% over the same period. Despite reductions in industrial, residential and commercial contributions, the forecast increase in road traffic is the cause of this trend in CO_2 emissions. Road transport emissions are projected to rise by almost 39%, from 3.1 megatonnes in 1990 to 4.3 megatonnes in 2010.

The validity of these projections of CO_2 can be checked by comparing with national forecasts which also suggest a reduction in emissions from the industrial sector and increases from the transport sector albeit, not as large (approximately 30% over the same period). However, the geographical location of Kent, with strategic transport corridors from continental Europe to the UK, is reflected in the greater proportion of total emissions contributed by this sector (40-42%) compared to approximately 23% nationally.

Conclusions

To complete this study, a comprehensive review of emission inventory evaluation techniques was undertaken and used. Whilst accepting that no one evaluation technique is correct, the results suggest a high degree of confidence in the inventory data enabling its use in the development of appropriate land use and transportation policies across the study area. The use of tolerances consistent with the limited amount of literature available yields uncertainty estimates of emissions similar to the NAEI data. However, this approach relies on 'expert judgement' in assigning such tolerances with no clear guidance. Evidently, further work is required to determine the robustness of this expert judgement through systematic study and reference to measurements in the field.

The pattern of NO_x and PM_{10} emissions in Kent and Medway are in line with national data. Emissions of these two pollutants have declined significantly since 1990 and are projected to continue to decline up to 2020. The most significant reduction is from the transport sector. This is in contrast to CO_2 emissions with the trend in reducing emissions expected to reverse from 2010. The most significant contributor is from the transport sector.

Disclaimer

The views expressed in this paper are those of the authors and not necessarily of the organisations they work for.

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Table 1
Techniques for Verification of an Emissions Inventory

Verification Technique	Elements	This Study
Documentation of data and Procedures	Details of the selection of procedures, assumptions and factors	A critical evaluation of each estimate was made in accordance with LRC 'best practice' (Hutchinson and Clewley, 1996, DETR, 2000j) *and Approach 1;* Murthy *et al* (1990).
		An *independent review* was undertaken by a third party for part qualification as registered auditor with the Institution of Environmental Management and Assessment (Critten, 2001).
	Data quality objectives	The emissions inventory was subject to quality assurance procedures (ISO 9001) as part of Mott MacDonald project management requirements.
Quality checks in relation to the application of data	Basic applications as a verification technique	The emissions data were checked in terms of providing, in combination with dispersion modelling, estimates of annual mean NO_2 and PM_{10} concentrations on a local to regional scale. The inventory provides a reasonable disaggregation of emissions by sector over the study area. The spatial resolution of the inventory does not allow for very local scale (ie. tens of metres) assessment.
Comparison of alternative estimates	Consideration of alternative methods and data: Comparisons of emissions densities and factors	Reference to national emission factors where applicable. *Completeness check* for NO_x and PM_{10} with reference to the urban inventories. Consistency check for NO_x and PM_{10} by comparing emissions with underlying geography (eg. increased emissions with urban density) and with local NAEI extract and urban inventories. Comparison of CO_2 emissions with national inventory.
Uncertainty estimates	Spatial and temporal issues as well as overall magnitude of emissions	Spatial comparison of NO_x and PM_{10} emissions with local NAEI extract; Temporal evaluation by comparing annual trends with observed data / national estimates: Lindley and Longhurst (1996); Lindley *et al* 1999, 2000; Sadler (1998); and Niemeier, Lin & Utts (1999).
	Sensitivity studies	*Data quality ratings* provided for NO_x and PM_{10} as per Sadler (1998) Table 1. Parameter sensitivity considered in terms of results being highly sensitive to one parameter, implying the estimate is less valid: Buckland and Middleton (1999). Specific *checks on systematic error and bias* not undertaken. Monte Carlo analysis undertaken to identify bounds of confidence.
Ground truth verification	Survey analyses	*Random sampling and remote measurement* not undertaken.
	Monitoring analyses	Comparison of observed trends in ambient levels of NO_x and PM_{10}, and trends in emissions: Sturm *et al* (1999); Lindley *et al* (1996).
		Evaluation of the emissions inventory and dispersion model combined, with reference to observed ambient concentrations; in accordance with common practice (eg. Renolds and Broderick, 2000) and *Approach 2;* Murthy *et al* (1990).
	Source sampling	Comparison of observed NO_x / PM_{10} ratios and emissions: Funk *et al* (2001).
	Measurement of operating parameters	*Receptor modelling* not undertaken.

Note: Table Headings *Verification Techniques and Elements;* from Lindley *et al* (2000)

Table 2
Summary Comparison of Emission Inventories

Inventory	Area (km^2)	Population (millions)	NO_x (kTonnes)	PM_{10} (kTonnes)	CO_2 (kTonnes)
This Study (1997)		1.56	101.4	7.911.6	8864
This Study (2000)		1.6	94.8	7.2	9190
NAEI (Kent and Medway) (1998)	3908	1.575	[6]40.145	[6]2.847	nr
NAEI (UK) (1998)	[2]244880	5659	1835	184	546000
Bristol (1996)	181	0.402	10.152	1.162	1995
Glasgow (1996)	239	[3]0.6	11.894	1.023	2721
Gr. Manchester (1995)	1552	2.577	75.434	11.188	21486
London (1995)	2466	[4]7.8 (7.187)	147.581	9.844	34243
Merseyside (1996)	1009	1.409	44.404	14.332	12973
Middlesbrough (1996)	261	0.145	32.314	2.226	20477
Southampton & Portsmouth (1996)	465	0.406	23.012	2.147	4007
Swansea & Port Talbot (1996)	358	0.368	15.759	2.312	2463
West Midlands (year not reported)	899	[5]2.5 (2.62)	46.519	4.021	28050
West Yorkshire (1996)	655	2.113	24.79	2.003	9704

nr not reported
[1] population data from National Statistics (2001) unless otherwise stated
[2] from National Statistics (2001)
[3] data from Glasgow City Council website
[4] data from Buckingham C, Sadler L and Shah S (1998)
[5] data from Hutchinson and Clewley (1996)
[6] excludes Part A emissions
other data from NAEI (DETR, 1998), LRC urban inventories and this work.

(Source : this study)

Table 3
Comparison With NAEI Data for Kent (excluding Part A Emissions)

Inventory	NO_x (ktonnes)	PM_{10} (ktonnes)
This Study (1997)	44.1	3.260
This Study (2000)	37.3	2.719
NAEI (Kent and Medway) (1998)	40.145	2.847

(Source : this study)

Table 4
Total PM_{10} Emissions by Sector (%)

	Part As	Part Bs	Other Industry	Major Roads	Minor Roads	Residential	Commercial	Shipping	Airports	Agriculture/ Construction
This Study 1997	58.8	5.3	-	30.5	4.9	<0.01	0	0.5	0.1	0.01
This Study 2000	62.4	5.7	-	26.7	4.7	<0.01	0	0.4	0.1	0.01
NAEI (Kent and Medway 1998)	0.32 (excl Part As)			1.2		0.7	0.7			
NAEI (UK 1997)	54			23		15	-	1	0	-
Bristol	52.7			35.5		1.6		-	-	-
Glasgow	11.8			73.2		4.5		-	-	-
Gr. Manchester	24.9			31.3		43.5		-	-	-
London	9.9			77.3		2.4		-	0	-
Merseyside	7.7			90.1		0.7		-	-	-
Middlesbrough	73.4			19.1		0.1		-	-	-
Southampton and Portsmouth	58.4			23.9		11.7		-	-	-
Swansea and Port Talbot	80.1			12.9		5.4		-	-	-
West Midlands	27.7			55.9		8.5		-	-	-
West Yorkshire	28.5			63.7		1.5		-	-	-

Notes: NAEI (UK) data for 1997 from DETR (2000b); Part A data not available
NAEI (Kent and Medway) for 1998
Urban Inventory Data: for Merseyside, Bristol and Southampton/Portsmouth: Buckingham *et al* (1997); for West Yorkshire, Glasgow, Swansea / Port Talbot and Middlesbrough: Buckingham *et al* (1998); for West Midlands: Hutchinson and Clewley (1996); for Greater Manchester: Buckingham *et al* (1997); for London: Buckingham *et al* (1997)

(Source : this study)

Table 5
Assumed Tolerances in Emissions Data Estimates (for Monte Carlo Analysis)

Source	Quality Rating (0-3)	Tolerance (+/- %)	Comment
Part A Processes	1-3	35	Data reported and verified by the Environment Agency. Sturm *et al* (1999) suggests ±35% based on measurement data
Part B Processes	1-2	100	Data generally unreported so based on typical emissions by process type. Sturm *et al* (1999) suggests ±65% based on heating capacity. Additional tolerance added for uncertainty in activity data.
Other Industry	0-1	100	30% tolerance estimated for NAEI; Sturm *et al* (1999) suggests ±65% based on heating capacity. Additional tolerance included for spatial error
Major Roads	3	50	DETR (1999g) does not provide estimates of uncertainty. Vogel *et al* (2000) suggest a difference of ±20% between estimated and measured vehicular emissions; Bull and Zimmann (1997) suggest a typical factor of four between emissions models; Mensink *et al* (2000) report a range of ±80% between measured hot NO_x emissions; Kuhlwein and Friedrich (2000) suggest a statistical error for hot NO_x emissions of ±35% (at 68.3% confidence limits); El-Fadel and Hashisho (2000) suggest ±65%, John *et al* (1999) consider greater than ±10 to 20%, Reynolds and Broderick (2000) and Colvile *et al* (2001) suggest a global factor of two. ±50% selected as a compromise between reported values.
Minor Roads	2-3	75	See discussion for Major Roads; additional tolerance added for spatial and activity data error.
Shipping	3	50	30% tolerance estimated for NAEI, additional tolerance included for spatial error and comments by Sadler (1998)
Airports	1-2	50	30% tolerance estimated for NAEI, additional tolerance included for spatial error and comments by Sadler (1998)
Residential Buildings	2-3	50	30% tolerance estimated for NAEI, additional tolerance included for spatial and activity data error and comments by Sadler (1998)
Commercial Buildings	1-2	50	30% tolerance estimated for NAEI, additional tolerance included for spatial and activity data error and comments by Sadler (1998)
Agriculture	0-2	50	30% tolerance estimated for NAEI, additional tolerance included for spatial error and comments by Sadler (1998)

Notes: Quality rating based on aggregate of emissions factor quality and availability of activity data

Table 6
Applied Tolerances in Annual Mean Emissions by Year

Year	Tolerance (%)
1990	±10 %
1997	±0 %
2000	±0 %
2005	±5 %
2010	±10 %
2020	±20 %

(Source : this study)

Table 7
Range of Uncertainty in Annual Mean Emissions

Year	NO_x	PM_{10}	CO_2
1990	± 50%	± 53%	± 43%
1997	± 39%	± 43%	± 45%
2000	± 38%	± 41%	± 46%
2005	± 40%	± 43%	± 49%
2010	± 43%	± 44%	± 52%
2020	± 51%	± 53%	± 57%

Note: Range of uncertainty expressed as the total for all source categories. See text for methodology.

(Source : this study)

Table 8
Comparison of NO_x : PM_{10} Ratios of Observed Concentrations and Estimated Emissions

Monitoring Site (1997-2000)	Class	Annual Mean NO_x : PM_{10} Ratios	
		Observed	Emissions
Folkeston Suburban	b	1.8	38
Luton Background	b	3.3	4.2
Maidstone Rural (Detling)	b	1.5	45
Sevenoaks Background (Greatness)	b	2.5	35
Stoke Rural	b	1.8	38
Gravesham Ind Background (Northfleet)	i	1.1	6.9
Chatham Roadside (A2)	r	4.8	52
Dartford Roadside (St Clements)	r	2.0	2.9
Gravesham Roadside (A2)	r	4.6	2.1
Maidstone Roadside (Fairmeadow)	r	5.7	48

(Source : this study, *after* Funk *et al*, 2001)

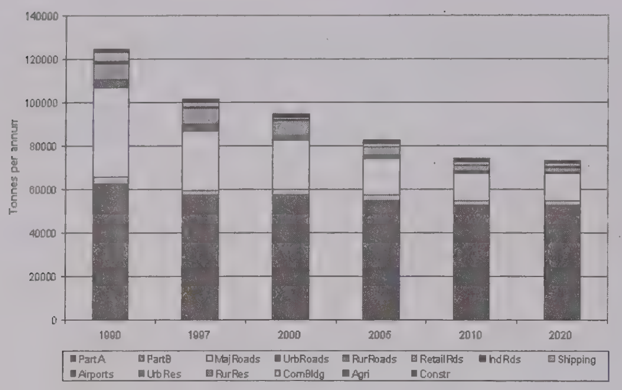

Figure 1. NO_x Emissions: 1990 to 2020

(Source: this study)

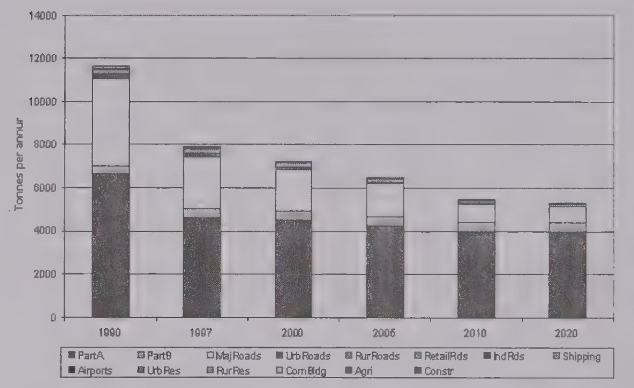

Figure 2. PM_{10} Emissions: 1990 to 2020

(Source: this study)

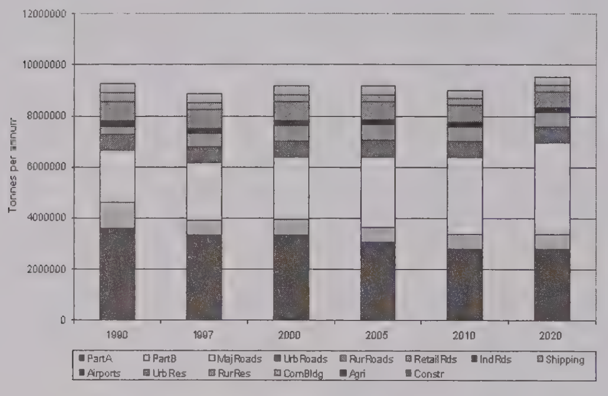

Figure 3. CO_2 Emissions: 1990 to 2020

(Source: this study)

Factors Affecting Inter-annual Variability of NO_x and NO_2 Concentrations from Single Point Sources

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1. Introduction

In the application of dispersion modelling for regulatory purposes questions frequently arise regarding the meteorological data used for the study: in particular how many years data are sufficient; is the meteorological site representative of the dispersion site? Although studies have been conducted on the impact of meteorology from different years on ground level concentrations from point sources[1,2] these studies have generally focused on the source contribution, without consideration of background concentrations or of any chemical reactions that may take place. This approach is sufficient for releases of SO_2 from large dominant sources. However, most regulatory studies now focus on NO_2, for which background concentrations and chemical reactions are important. Investigation of the inter-annual variability of only the process contributions to NO_x may give little indication of what is actually important, that is, the inter-annual variability of the NO_2 concentration.

In this paper we present some calculations of NO_x and NO_2 using ADMS 3.1[3], in which the impacts of both background pollutant concentrations and chemical reactions are assessed. The focus is on isolated point sources without building effects. The impact of the choice of meteorological site is not considered here.

2. Model Input Data

2.1 Source Data

Two 'standard' sources were modelled, a typical power station and a buoyant release from a height of 50m. The source details are given in Table 1. The NO_x emissions are assumed to be 10% NO_2 by volume.

2.2 Met Data

Five years of sequential met data, from 1992-96, from Birmingham (Elmdon) were used. Table 2 gives the mean wind speed at 10m for each year. This can be seen to vary by up to about 20% from year to year.

Table 1 – Source Data

Source	Power station	50m source
Height (m)	199	50
Diameter (m)	13	1
Exit velocity (m/s)	22	9
Temperature (°C)	130	200
Emission rate NOx (g/s)	1000	15

Table 2 – Mean Wind Speeds

Year	Mean wind speed (m/s)
1992	3.8
1993	3.8
1994	4.2
1995	3.6
1996	3.4

2.3 Background Data

Background concentrations of NO_x, NO_2 and O_3 from Lullington Heath (a rural monitoring site) from 1992-96 were used. Harwell monitoring site, which is nearer to Elmdon meteorological station was not used as there were insufficient data for the years in question.

If data for any pollutant were missing for a particular hour, values for all 3 pollutants were replaced by the average values over all 5 years. This was necessary for 21% of hours in 1992, 26% in 1993, 38% in 1994, 7% in 1995 and 15% in 1996.

Statistics of the background data are given in Tables 3 and 4. The annual mean values show little variation from year to year. There is more variation in the high percentiles – in particular the maximum hourly concentration of NO_x and NO_2 and the 99.9[th] percentile of NO_x were unusually high in 1992.

Table 3 – Statistics of Hourly NO_x and NO_2 Background Data (ppb)

	1992		1993		1994		1995		1996	
	NO_x	NO_2	NO_x	NO_2	NO_x	NO_2	NO_x	NO_2	NO_x	NO_2
Annual mean	9.2	7.5	10.0	8.5	9.9	8.4	8.7	7.5	10.2	8.3
98[th] percentile	40	30	39	30	35	30	32	25	43	29
99.8[th] percentile	162	46	92	42	81	58	71	37	80	40
Maximum	311	90	102	53	101	64	93	43	109	46

Table 4 – Statistics of Hourly O_3 Background Data (ppb)

	1992	1993	1994	1995	1996
Annual mean	31	28	29	30	27
98[th] percentile	68	60	62	77	59
99.9[th] percentile	99	96	102	122	85
Maximum	104	117	117	134	90

3. Modelling Methodology

Two chemistry schemes were used – the GRS (Generic Reaction Set) and the OLM (Ozone Limiting Method).

3.1 GRS Chemistry Scheme

The GRS chemistry scheme[4] is a semi-empirical photochemical model which reduces the complex set of

reactions between NO, NO_2, O_3 and VOCs (volatile organic compounds) to just a few reactions. The two reactions involving NO, NO_2 and O_3 are:

$$NO_2 + h\nu \rightarrow NO + O_3 \quad (1)$$

$$NO + O_3 \rightarrow NO_2 \quad (2)$$

In addition there are parameterisations of reactions involving ROC, RP, SGN and SNGN which represent reactive organic compounds, radical pools, stable gaseous nitrogen products and stable non-gaseous nitrogen products respectively. $h\nu$ represents ultra-violet radiation.

In this illustrative case we are not considering emissions or background concentrations of VOCs and thus the GRS reduces to equations (1) and (2) only. This is the NO_X chemistry scheme included in ADMS 3. The inputs to the chemistry scheme are the sum of the process contribution, the background concentration and the ultra-violet radiation which is estimated hour by hour. The chemistry model assumes that ambient air mixing into the plume as it is transported downstream is uniformly mixed across the plume. In general this assumption is likely to lead to a small overestimate of NO_2 and a corresponding underestimate of O_3 for a given NO_x concentration.

3.2 OLM Chemistry Scheme

The OLM scheme[5] estimates NO_2 concentrations by assuming NO reacts instantaneously with all available ozone via a single reaction:

$$NO + O_3 \rightarrow NO_2 + O_2$$

For each hour, the process contribution to the NO_2 concentration is calculated using

$$[NO_2]_{OUT} = [NO_2]_{IN} + \text{Minimum} ([NO]_{IN}, [O_3]_{IN})$$

where all concentrations are in ppb, and the input concentrations $[NO_2]_{IN}$ and $[NO]_{IN}$ are the process contribution from the source. After this calculation has been carried out, a background NO_2 concentration suitable for the current hour is added to the concentration. This scheme has been implemented in ADMS 3 for the purposes of the current study and is also included in some US EPA dispersion models. The scheme is likely to be very conservative since the reaction is assumed instantaneous, there is no reverse photochemical reaction (the dissociation of NO_2) and, as in the standard version of ADMS, mixing is assumed uniform. The magnitude of this conservatism is considered below.

3.3 Model Calculations

ADMS 3.1 was used to calculate concentrations for each hour in the period 1992-1996 for each of the cases (1) to (6) below.

Case (1) Without chemistry, no contribution from background (i.e. process contribution only)

Case (2) With GRS chemistry using the appropriate year of background data

Case (3) With GRS chemistry always using 1992 background data

Case (4) With GRS chemistry always using 1996 background data

Case (5) With GRS chemistry always using 1994 met data

Case (6) With ozone limiting (OLM) chemistry using the appropriate year of background data

The hourly average concentrations were calculated on a grid of 31 by 31 points (20km square for the power station, 2km square for the 50m source) and used to derive annual average, 98th and 99.9th percentile and maximum hourly average concentrations of NO_X and NO_2.

4. Model Results

Results shown here are the maximum ground level concentrations for each statistic calculated. Results are presented for each of the six scenarios listed above, and for the case of no process contribution i.e. assuming only background concentrations are present.

Firstly, results of NO_X and NO_2 are shown as graphs of the maximum ground level concentration against the year (Figures 1-6). The following should be noted:

- NO_X results for the ozone limiting method are not shown, as these are identical to the GRS chemistry results, since both chemistry methods conserve the total volume of NO_X.
- For the power station, the maximum ground level 98th and 99th percentile concentrations were found to be almost identical to the background concentrations, hence these graphs are omitted. The graphs of annual average and maximum hourly average concentrations are shown in Figures 1 and 2.
- For the 50m source, graphs are shown for the annual average, 98th and 99th percentile and maximum hourly average concentrations (Figures 3-6).

Secondly, the mean and standard deviation of each statistic are given, for each pollutant, for each source, in Tables 5-8. Table 9 shows the percentage of the process NO_X contribution which is NO_2 at the location of the maximum ground level concentration. This illustrates the impact of local chemistry, as modelled by the two chemistry schemes, on NO_2 concentrations.

5. Discussion

5.1 Process Contribution Study

Considering first the process contribution only (Case (1)), it can be seen that the relative contribution by the process to the total concentration is greater for the 50m source than for the power station. It can also be seen that the relative contribution from the process to the total concentration is smaller for the annual average (the mean NO_X concentrations are 0.4ppb for the power station compared with the

background of 9.6ppb) than for the percentiles (the maximum hourly NO_X concentrations are 57ppb for the power station compared with the 143ppb background). This is true for both sources and both pollutants, NO_X and NO_2.

In this case process contributions of NO_x from the 50m source are greater in absolute terms than those from the power station although, of course, the power station has an impact over a much greater geographical area than the 50m source.

5.2 Total Concentration – Process Plus Background

Comparison of impacts of inter-annual variation of background concentrations with those of meteorology

The annual mean concentrations including background concentrations and chemistry effects are dominated by the background concentrations. The inter-annual variation in the background concentration causes more variation in the results than the inter-annual variation in the meteorology. In Figures 1 and 3, the results for the cases where the same background data are used for every year, Cases (3) and (4), show the least variation. The case where the background values used varies from year to year, Case (2), shows the greatest variation. This trend is confirmed by the tabulated values of standard deviation. The standard deviation is smaller for Cases (3) and (4) where the same background is used for all years than for Case (2). The difference is most pronounced for the power station where the background dominates the ground level concentrations whereas for the 50m stack, for which the relative contribution from the process is smaller, variations of both meteorology and background are important.

Comparison of GRS and OLM chemistry schemes

For the power station, the NO_2 annual mean results for the OLM scheme are similar to the GRS scheme due to the low process contribution from the power station in comparison with the background concentration. For the 50m source the OLM scheme predicts higher mean NO_2 concentrations than the GRS scheme as here the process contribution is higher compared with the background concentration.

Comparing Case (2) (GRS chemistry), Case (6) (OLM chemistry) and Case (5) (GRS chemistry with constant meteorology), it can be seen that the differences in NO_2 results between the GRS and OLM schemes are at least as large as the variations due to meteorology.

Composition of process contribution at location of maximum concentration

Table 9 shows the percentage of NO_x (by volume) present as NO_2 at the point of maximum concentration for the Case (2) using GRS and Case (6) using OLM. The Table shows annual averages for each year and for one case a high percentile (99.9th).

For the 50m source the GRS scheme predicts between 47% and 58% of the annual average process contribution is NO_2 whereas OLM predicts between 89% and 95%. For the power station the GRS scheme predicts between 46% and 100% is NO_2 whereas OLM predicts over 98% is NO_2. The unrealistic oxidation of NO predicted by OLM, leading to over-predictions of NO_2, is most apparent for the 50m source.

This comparison shows the inadequacy of assuming that 50% of the process contribution is NO_2 at the location of maximum concentration as is often used, supposedly to give a conservative estimate of oxidation to NO_2. The percentage NO_2 depends on the averaging time, the percentile and the process contribution to total NO_X.

6. Conclusions

(i) Although it is useful to model the process contribution to NO_x, this may give little indication of either the NO_x or NO_2 climate (total concentration) if the relative contribution of the process contribution is small.

(ii) For model calculations of NO_2 it is important to consider the background concentrations and their inter-annual variation. This has more impact on predicted concentrations than inter-annual variations in meteorology.

(iii) In a modelling study the choice of chemistry scheme or neglect of chemical reactions is at least as significant as the inter-annual variation in meteorology.

(iv) The Ozone Limiting Method (OLM) leads to large overestimates of NO_2 when the process contribution is significant compared with the background concentration.

(iv) Representing the process contribution to NO_2 as a fixed percentage of NO_X, e.g. assuming 50% of the process NO_X is NO_2, leads to a poor estimate of the process contribution to NO_2. It may result in either over-estimates or under-estimates of total NO_2.

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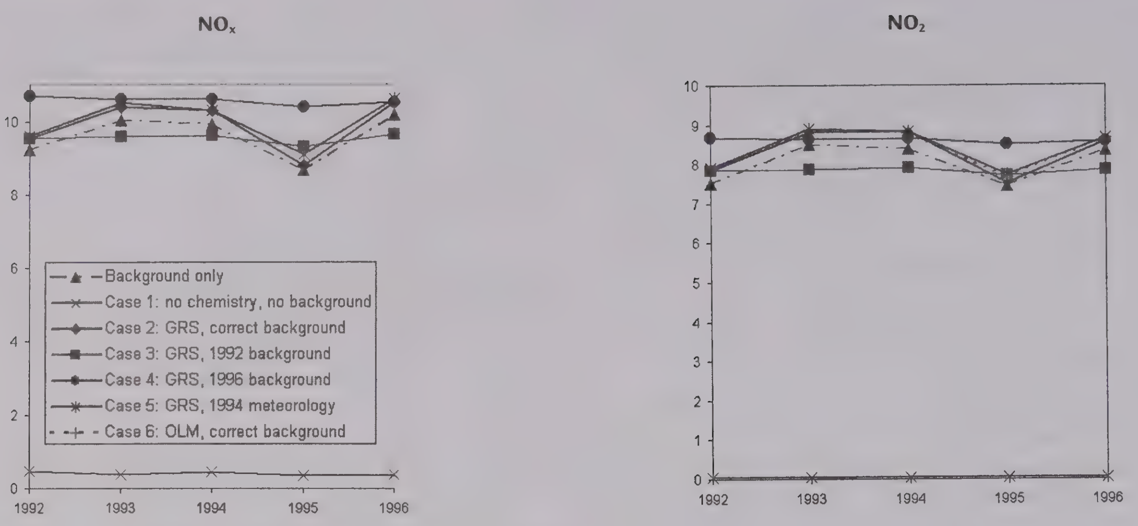

Figure 1 – Power station, maximum annual mean concentrations (ppb)

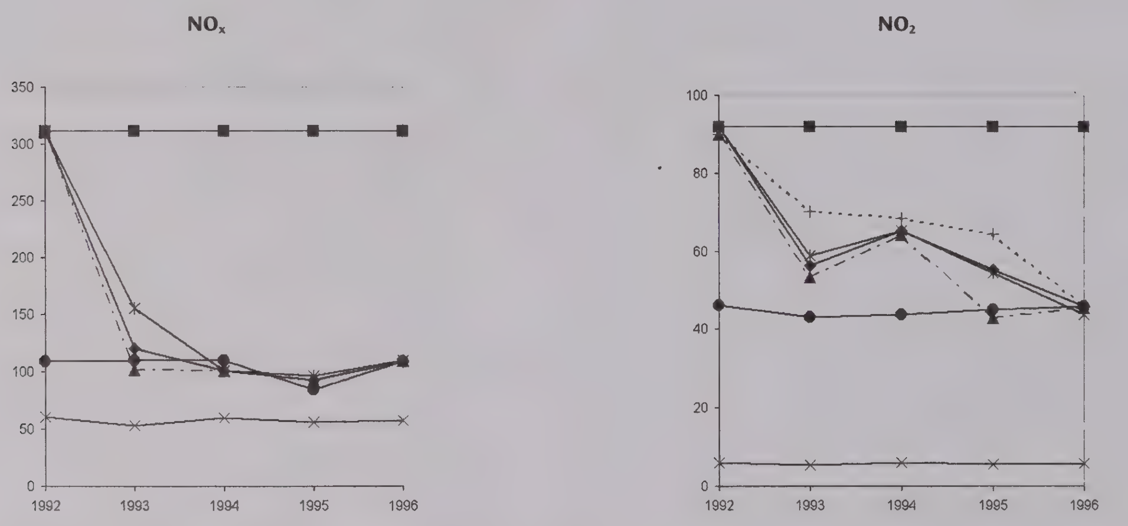

Figure 2 – Power station, maximum hourly average concentrations (ppb)

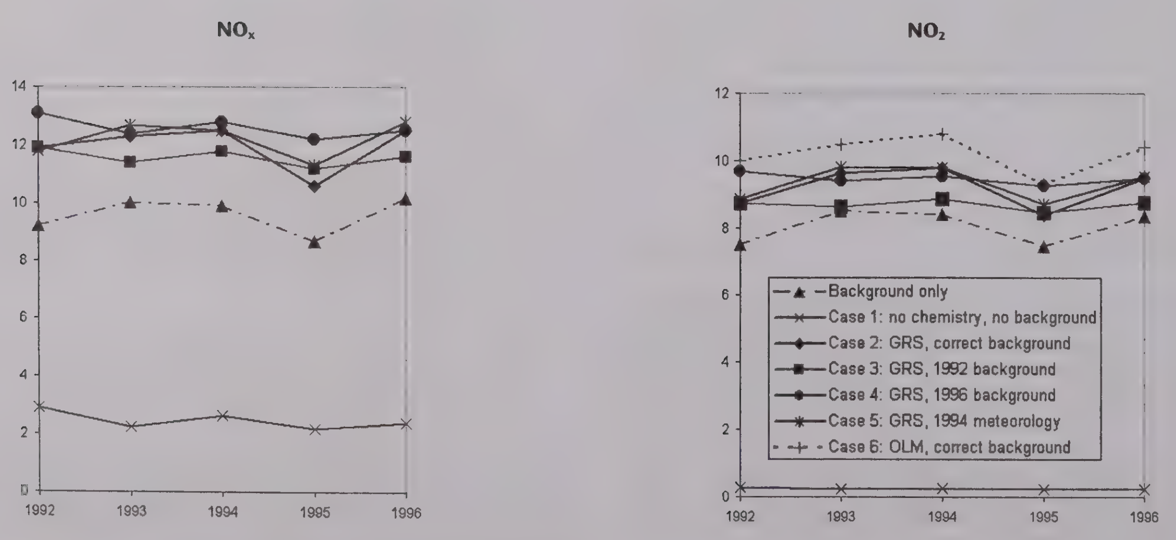

Figure 3 – 50m source, maximum annual mean concentrations (ppb)

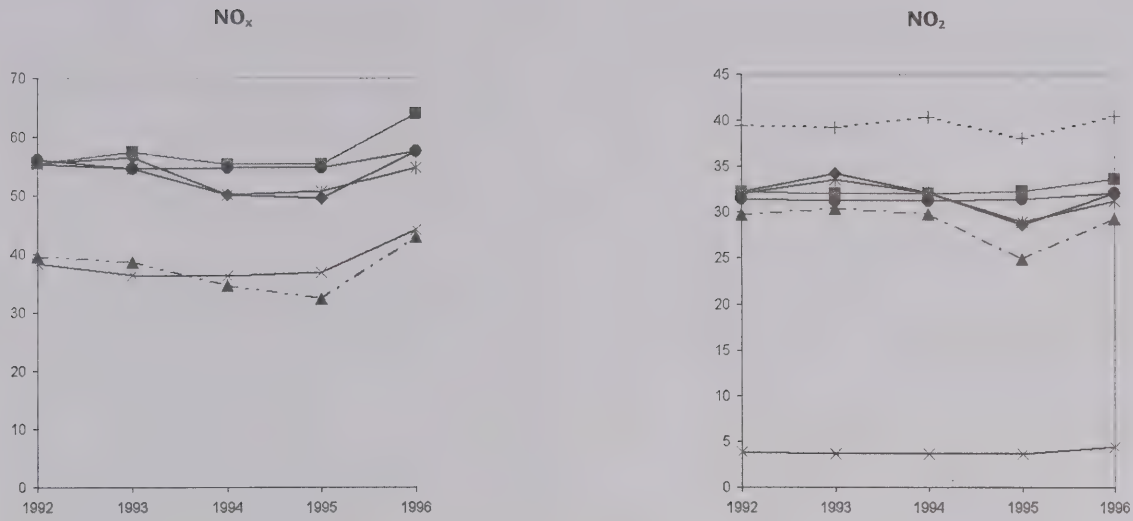

Figure 4 – 50m source, maximum 98th percentile concentrations (ppb)

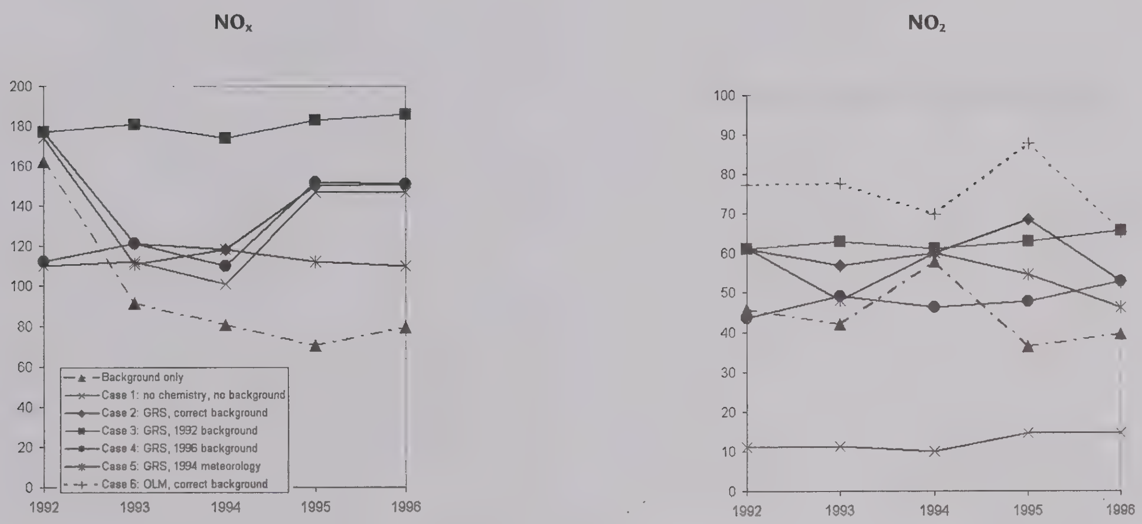

Figure 5 – 50m source, maximum 99.9th percentile concentrations (ppb)

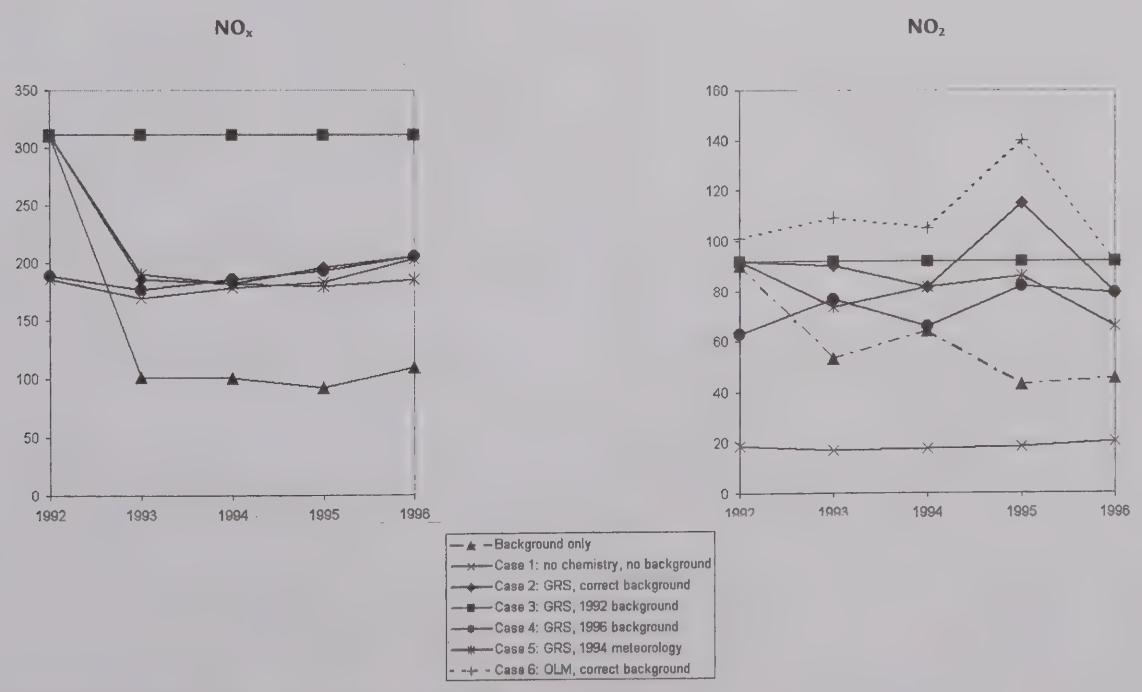

Figure 6 – 50m source, maximum hourly average concentrations (ppb)

Table 5 – Power Station, mean and standard deviation of maximum NO_x results (ppb)

	Background only		Case 1 No chemistry, no background		Cases 2 & 6 GRS or OLM, correct background		Case 3 GRS, 1992 background		Case 4 GRS, 1996 background		Case 5 GRS, 1994 meteorology	
	Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std
Annual mean	9.6	0.57	0.4	0.05	9.9	0.65	9.5	0.12	10.6	0.10	10.0	0.58
98th percentile	38	3.75	10	1.65	38	3.60	40	1.06	43	0.37	39	3.26
99.9th percentile	97	33.2	29	2.36	95	30.7	153	1.20	79	1.06	97	27.7
Maximum	143	83.9	57	2.81	147	82.6	311	0.00	104	10.2	155	80.9

Table 6 – Power Station, mean and standard deviation of maximum NO_2 results (ppb)

	Background only		Case 1 No chemistry no background		Case 2 GRS, correct background		Case 3 GRS, 1992 background		Case 4 GRS, 1996 background		Case 5 GRS, 1994 meteorology		Case 6 OLM, correct background	
	Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std
Annual mean	8.1	0.46	0.04	0.01	8.3	0.53	7.8	0.07	8.6	0.06	8.4	0.49	8.4	0.5
98th percentile	28.8	2.00	1.0	0.17	45.0	2.15	30.2	0.47	29.5	0.07	29.5	2.01	45.5	7.12
99.9th percentile	44.5	7.43	2.9	0.24	57.5	7.18	46.3	0.73	40.0	0.04	45.3	6.85	60.1	14.1
Maximum	59.2	17.1	5.7	0.28	62.9	15.7	91.9	0.00	44.7	1.16	62.8	16.2	67.7	14.1

Table 7 – 50m source, mean and standard deviation of maximum NO_x results (ppb)

	Background only		Case 1 No chemistry, no background		Cases 2 & 6 GRS or OLM, correct background		Case 3 GRS, 1992 background		Case 4 GRS, 1996 background		Case 5 GRS, 1994 meteorology	
	Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std
Annual mean	9.6	0.57	2.5	0.26	12.0	0.71	11.6	0.26	12.6	0.32	12.2	0.58
98th percentile	38	3.73	123	19.3	143	21.8	180	4.26	129	18.6	125	24.7
99.9th percentile	97	33.2	168	12.2	201	47.4	295	1.20	176	9.35	195	51.1
Maximum	143	83.9	184	11.2	216	48.1	311	0.00	190	9.50	210	50.7

Table 8 – 50m source, mean and standard deviation of maximum NO_2 results (ppb)

	Background only		Case 1 No chemistry no background		Case 2 GRS, correct background		Case 3 GRS, 1992 background		Case 4 GRS, 1996 background		Case 5 GRS, 1994 meteorology		Case 6 OLM, correct background	
	Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std	Mean	Std
Annual mean	8.1	0.46	0.25	0.03	9.2	0.55	8.7	0.13	9.5	0.14	9.4	0.47	10.2	0.52
98th percentile	28.8	2.00	12.3	1.93	60.0	5.16	62.8	1.75	48.1	3.07	54.2	6.02	75.7	7.74
99.9th percentile	44.5	7.43	16.8	1.22	84.1	15.2	87.5	1.78	64.1	5.69	71.3	8.89	107	17.2
Maximum	59.2	17.1	18.4	1.12	91.6	12.6	91.9	0.00	71.8	7.80	79.8	9.21	114	15.4

Table 9 – NO_2 process contribution expressed as percentage of NO_x process contribution. Figures are annual average except those bracketed which are 99.9th percentile.

	% process NO_x that is NO_2			
	Case (2) GRS		Case (6) OLM	
Year	Power station	50m source	Power station	50m source
1992	100	48	100	93
1993	82	55	100	92
1994	100	58	98	92
1995	46 (< 1)	47 (66)	100 (< 1)	95 (100)
1996	73	54	100	89

NO_x Chemistry in Power Station Plumes

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Introduction

The two most significant sources for oxides of nitrogen within the UK are road transport and power generation, accounting for an estimated 48% and 21% of 2000 UK emissions respectively (NAEI website).

The major proportion at the point of emission from both sources is in the form of nitric oxide (NO), and this comprises around 95% by volume of emissions from coal-fired power stations, and the remainder as predominantly nitrogen dioxide (NO_2).

After emission, oxidation of NO can occur, forming further nitrogen dioxide, which is the species of potential concern with respect to human health and this species is the subject of air quality standards, such as those given in the Air Quality Strategy (DETR, 2000). The target is for no location to record an annual mean hourly concentration of greater than 21ppb, or a 99.8th percentile hourly mean concentration greater than 105ppb by the end of 2005.

Since power stations contribute only about 20% of national emissions of NO_x, ground-level concentrations are frequently dominated by other sources such as local traffic and domestic emissions. Identifying the contribution of a local power station to measured NO_x concentrations is therefore difficult except, perhaps, during a few higher concentration peaks, which occur from time-to-time.

Since NO_2 is largely produced by a complicated secondary process after emission, it is not straightforward to predict through standard dispersion modelling. Pollutants are treated as passive quantities whose predicted ground level concentration will be proportional to the emission rate. This works for SO_2 and NO_x but not for NO_2. The simplest recourse for the modeller is to resort to assuming all NO_x might be converted to NO_2. However, this is potentially very conservative and may exaggerate the impact of a source on air quality.

This paper is based on research undertaken by the Power Generators through their Joint Environmental Programme (JEP) which has attempted to quantify the likely contribution from power station sources to NO_2 ground level concentrations on a short and long-term basis.

Plume NO_x Chemistry

It has been noted above, that the majority of power station emissions of NO_x are in the form of NO, which is subsequently oxidised in the troposphere by reaction with O_3. Tropospheric O_3 is formed by the oxidation of volatile organic compounds and carbon monoxide, or the dissociation of NO_2, both processes promoted by the presence of sunlight. As a result, its concentration varies considerably with both location and time of day and year, with the highest UK concentrations observed during summer afternoons in rural locations influenced either by air from major cities or from continental Europe.

The most important oxidation reaction under typical atmospheric conditions is the oxidation of NO by O_3:

$$NO + O_3 \rightarrow NO_2 + O_2 \quad (2.1)$$

The rate of reaction is very rapid, however the ultraviolet component of sunlight can cause the reverse reaction:

$$NO_2 + \text{sunlight } (h\nu) \rightarrow NO + O \quad (2.2)$$

(which combines with O_2 to form O_3)

This leads to a state of photostationary equilibrium during the day, with the position of equilibrium dependent upon sunlight intensity. At night no photo-dissociation occurs and oxidation continues until either nitric oxide or O_3 is exhausted.

Under conditions where this is the only significant oxidation pathway, the concentration (in ppb) of secondary NO_2 will not exceed the lower of the two concentrations, NO and available O_3 (also in ppb).

Direct oxidation by reaction with molecular oxygen can also occur via the termolecular reaction:

$$2NO + O_2 \rightarrow 2NO_2 \quad (2.3)$$

however the rate is proportional $[NO]^2$ hence its significance declines rapidly with decreasing concentration of NO. It is generally recognised as only of significance in extreme urban pollution episodes with NO_x concentrations above 500ppb, which is significantly higher than the ground level concentrations arising from power station emissions.

Hydroperoxy radicals (HO_2) and organic peroxy radicals (RO_2) are generated in the troposphere by oxidation of carbon monoxide or volatile organic compounds. These react with nitric oxide to form nitrogen dioxide and a further radical:

$$HO_2 + NO \rightarrow OH + NO_2 \quad (2.4)$$

$$RO_2 + NO \rightarrow RO + NO_2 \quad (2.5)$$

Carbon monoxide and volatile organic compounds are both associated with sources other than power stations, most notably traffic, and the reactions are relatively slow. This route for the generation of NO_2 is therefore not considered to be of direct relevance for power station plumes but contributes to the background O_3 concentration via the subsequent photo-dissociation of the NO_2. More detailed descriptions of the atmospheric processes can be found in PORG (1997).

Short-Term Impact of a Power Station Plume on NO_2 Concentrations

The ratio of NO_2 to NO in a power station plume will be influenced by several factors. As mentioned above, O_3 is the

primary oxidant involved in the formation of NO_2 in a plume and this must be mixed into the plume in order for the reaction to take place. Plume dilution by the process of dispersion is therefore a major factor in determining the formation of NO_2.

The contribution of plume concentrations to ground level concentrations is clearly going to be dependent on the fraction of any averaging period that the plume is in contact with the ground (or present at a monitoring site). This will be governed by the physics of the atmosphere – driven by meteorological processes. Large scale vertical motions and wind direction changes are the two main reasons for reducing the contribution of a plume during a given hour period.

Since the dissociation of NO_2 back into NO under the action of sunlight plays a significant role in determining NO_2 concentrations, time of day and time of year are also important factors to take into account when estimating NO_2 in a plume.

Finally, the background concentrations of NO and NO_2, as well as O_3 are important in determining the contribution of plume NO_x to NO_2 concentrations. This is because, once mixed intimately with the plume, there can be no distinction between plume NO_x and background NO_x.

The short-term AQS objective for NO_2 ground level concentrations is defined as 200 μgm^{-3} hourly average not to be exceeded more than 18 times in a year. In an assessment of the contribution of power station plumes to NO_2 at ground level it is therefore important to establish whether or not the plumes contribute significantly to NO_2 concentrations during periods when they exceed 200 μgm^{-3} (or 105 ppb).

Studies of monitoring data from sites around power station plumes demonstrate that high ground level concentrations of NO_x coincide with low background NO_x and NO_2. This is primarily because the conditions which cause the highest plume concentrations at the ground (very convective) are also those that disperse the low level sources of NO_x (cars etc) very effectively (Webb and Hunter 1998).

This means that an O_3 concentration at least close to that of the NO_2 standard (105 ppb) must be present in the background air in order to oxidise the requisite amount of NO.

Analysis of monitoring site data shows that such high ozone concentrations rarely occur (99.9th percentiles are typically 80-90 ppb, Webb and Hunter 1998).

The concentration of ozone present into which the plume is dispersing therefore sets the upper limit for the production of NO_2. However, in sunlight NO_2 can reconvert to NO, therefore the actual contribution of the plume to NO_2 will be significantly less than the ozone available during the day.

Evidence for exceedances of the NO_2 105ppb concentration was sought in 75 site-years of data from sites in the proximity of power stations. Of the 127 hours where this was exceeded, only 2 hours contained possible contributions from power stations to total NO_x of more than 25%. No exceedances were found with even a 50% contribution from local power stations. In all other cases the contribution to total NO_x was much less than 10%. 74ppb was the highest NO_2 concentration measured during any time when a power station was likely to have been the predominant source (Webb and Hunter 1998).

Figure 1 illustrates the progress of two contrasting plume-grounding episodes and provides a good illustration of several points pertinent to statistical analyses (Futter *et al* 2002).

The upper graph shows an incident at Jenny Hurn (North Trent valley) on 2nd and 3rd September 1999. This series of three occasions when the plume is detected, typifies the course of a summer episode. The wind was from the south on both days, with the temperature around 25°C. On the 2nd, the wind speed indicates a plume age of about two hours, with that on the 3rd around an hour older. All three plumes caused a sharp peak in SO_2 and NO_X concentrations superimposed on a low background. Total oxidant concentration can be seen to give a good representation of likely O_3 availability, following a smooth curve peaking in mid-afternoon. O_3 was visibly depleted during the plume periods, but not close to exhausted. This is due to the elevated oxidant (50ppb and 80ppb) combined with low background NO_X concentrations, and suggests that ozone limiting would not have occurred at ground level along the dispersing plume.

The lower graph is taken from measurements at Marton (North Trent valley) on 20th and 21st December 1997. This is an excellent example of plume grounding occurring during the winter with a limited supply of O_3 and the resultant complete ozone limiting. Elevated concentrations of NO_X were observed at all sites during mid-December 1997, as a winter episode of trapped traffic emissions accumulated. This consumed almost all the available O_3 and overcast, cool weather (5°C) mitigated against further generation of O_3. A south-westerly wind brought air from one of the power stations to Marton, leading to a severe plume grounding with SO_2 in excess of 220ppb and an increase in NO_x from 20ppb to 70ppb. However the lack of available O_3 led to none of the NO being converted to NO_2 and the NO_2 concentration remaining constant.

These examples show that ozone limiting occurs on some, but not all occasions where a plume causes elevated NO_X concentrations.

The conclusion of these JEP studies is that the likelihood that power station operation will cause an exceedance of the AQS short-term NO_2 objectives is effectively zero.

Estimating Annual Mean NO_2/NO_x Ratios

The proportion of NO_x in NO_2 form averaged over an annual period will depend on the relative contributions from high, moderate and low plume concentrations; and the background NO_x and O_3 concentrations at the time of the impacts. The O_3 concentration varies on diurnal and seasonal cycles with substantial variability. There is no simple theoretical method, therefore, to predict the appropriate NO_2/NO_x ratio that should be applied to convert NO_x predictions to NO_2 predictions.

Several methods can be used to estimate the overall proportion of NO$_x$ that would be expected to be in NO$_2$ form given a prediction of annual average NO$_x$ due to a power station. These include use of dispersion models with some level of chemistry included (ADMS, AERMOD), and analyses of monitoring data near power stations (using upper limit oxidant limiting arguments or incremental NO$_2$ and NO$_x$ methods).

The method reported here was based on an analysis of historical monitoring data using SO$_2$ as a marker for power station plumes and, of all the methods examined, seemed to provide the best estimates (Futter *et al* 2002).

Measured hourly mean NO$_2$ and NO$_X$ concentrations at each site in the Yorkshire and Trent JEP networks were corrected for background concentrations by averaging the measured concentrations at all sites in the local network judged not to be downwind of any local power station.

A ratio of three between the emission concentrations of SO$_2$ and NO$_X$, both expressed in ppm by volume, was assumed as typical for the local power stations. This provided an estimate of power station NO$_x$ and, using total NO$_x$ and NO$_2$ concentrations from sites defined as not downwind, NO$_2$ due to the power station was estimated assuming the same ratio of NO$_2$/NO$_x$ as measured.

The criteria for a site not being downwind was that the locally measured wind direction indicated that there were no local power stations in an upwind arc allowing -15° for wind shear with height and ±30° for wind variability. Thus, sites were classified as background only when all local power stations were outside an arc from -45° to +30° of the local wind direction. Data were excluded when the wind speed was <0.5ms^{-1} as assignment of wind direction is unreliable under such conditions.

Although the local power stations in Yorkshire and the lower Trent Valley dominate SO$_2$ emissions in their respective areas, there are other significant sources which can affect measured SO$_2$ concentrations. In addition to the estimates of NO$_X$ based on the total SO$_2$ concentration, therefore, the measured concentration of SO$_2$ was corrected for background. This "corrected" SO$_2$ concentration was then used to estimate the power station NO$_X$ contribution.

Figure 2 shows the results of the analysis as applied to 37 site-years of data from JEP networks. The NO$_2$/NO$_X$ ratio appropriate to annual mean power station contributions was approximately 75% but varied with the total NO$_X$ concentration. At annual mean total NO$_X$ concentrations due to all sources of 10ppb, the typical NO$_2$ conversion for power station annual mean contributions was about 80% decreasing to about 60% by total NO$_X$ concentrations of 45ppb.

Conclusions

The concentrations of NO$_2$ in power station plumes at ground level are limited by the concentration of ozone in the background air and constrained during the day by the photostationary state condition.

It is extremely unlikely that a power station plume could cause an exceedance of the AQS short term target concentration of 200 µgm^{-3} because of oxidant constraints and the anti-correlation between events of high background NO$_x$ and those due to an elevated source such as a power station. This is demonstrated through extensive analysis of monitoring data close to power stations.

The NO$_2$/NO$_X$ ratio appropriate to annual mean power station contributions was determined to be approximately 75% based on monitoring data analysis but was found to vary to some extent with the total NO$_X$ concentration. At annual mean total NO$_X$ concentrations of 10ppb, the typical NO$_2$ conversion for power station annual mean contributions was about 80% decreasing to about 60% for total NO$_X$ concentrations of 45ppb.

This work focuses upon the impacts around large coal fired power stations in rural areas, and would not be generally applicable to all large sources of NO$_X$. Whilst the same principles could equally be applied to other power station sources (such as gas turbines) or indeed other combustion sources, the specific circumstances of such other source types would also need to be considered.

Acknowledgements

This review paper makes extensive use of research undertaken by colleagues working under the JEP, in particular Gill Hunter, Alan Webb and David Futter.

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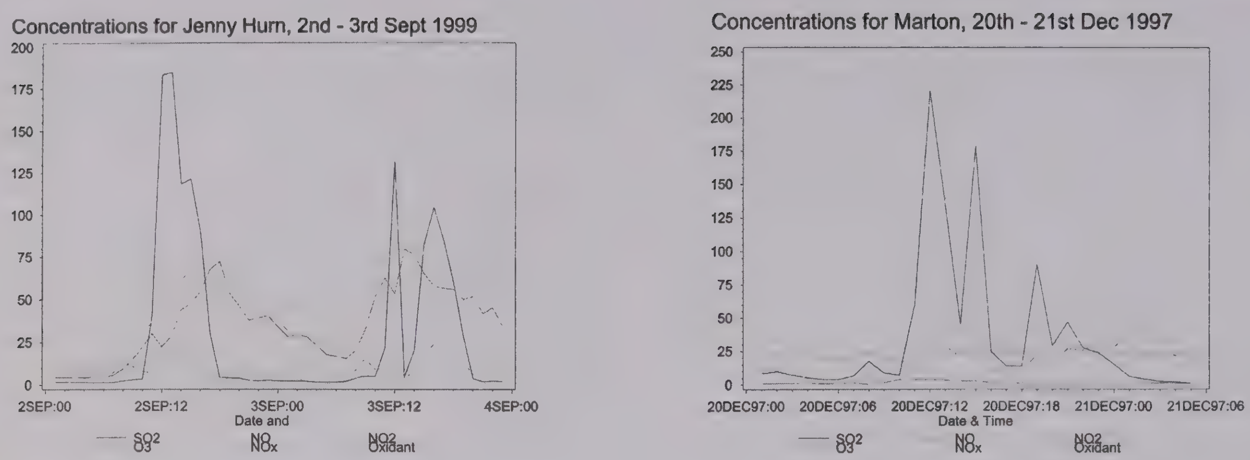

Figure 1

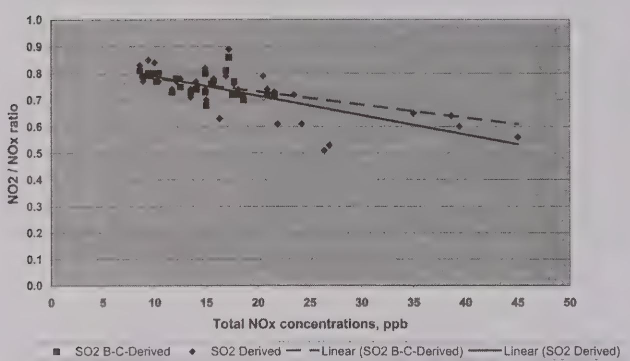

Figure 2

Air Quality Standards and Guidelines: Impact on Vegetation and Ecosystems, Current and Future Thresholds

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Introduction

In this paper an overview will be provided of current air quality standards and guidelines employed in the UK. These will be placed in a historical perspective, followed by a description of the methodology and processes by which they have been derived. Consideration will then be given to possible future changes in such standards and guidelines, on the basis of evolving research and changing air pollution climates.

Historical Perspective

Air pollution legislation and control policy has evolved over the last 150 years or so, following a cyclical pattern, as concerns change over the nature of the receptors to be protected against its adverse impacts. The earliest reports of damage to vegetation date back to the publication in 1661 of John Evelyn's famous treatise, *"Fumifugium: Or the Inconvenience of the Aer and Smoake of London Dissipated"*. Despite his pleas for effective coal smoke control measures, London's air quality continued to deteriorate well into the middle of the last century. However, it is worth noting that the first effective legislation in the world to control air pollution was enacted in 1863, with the establishment of the Alkali Inspectorate, with the specific remit to abate emissions of hydrochloric acid gas from alkali works in north-west England, which was devastating local vegetation.

Subsequent legislation and policy neglected effects on natural ecosystems, forestry and agriculture, until the late 1980s. This was despite strong evidence for serious problems, such as the destruction by air pollution of *Sphagnum* moss on the blanket bogs of the southern Pennines, problems of growth of commercial conifers and pasture grasses in the same area, and difficulties in growing plants which overwinter in a green state, in polluted urban areas. While problems of SO_2 and smoke-induced damage to vegetation declined dramatically with the sharp fall in concentrations of these pollutants in the latter part of the 20th century, other phytotoxic gases increased. In particular, nitrogen oxides rose, with increasing motor vehicle emissions and by the mid 1970s it was recognised that these were resulting in the production of O_3 concentrations, widespread across rural areas of much of the country, of sufficient magnitude potentially to harm vegetation.

Concerns over widespread problems of acidification arising from nitrogen and sulphur oxides and their products led to the establishment of the UNECE's Long Range Transboundary Air Pollution Convention with its subsequent protocols. Subsequently, concerns grew over the direct effects of the phytotoxic gases, O_3, SO_2, NO_x and NH_3 with appropriate protocols and legislation developed by both the UNECE and the European Commission. Also increasing evidence was produced of widespread adverse effects of nitrogen deposition on low nitrogen ecosystems, with moves to control emissions of its reduced, as well as its oxidised forms. Thus the UNECE has established critical levels and critical loads as a cost-effective receptor specific method of minimising ecosystem damage. The pioneering work of the UNECE has effectively been adopted in terms of its standards by other organisations; notably the EC, the World Health Organisation (WHO) and national governments in Europe. Possible future developments in terms of modifying these standards will be discussed later.

How Have Air Quality Standards/Guidelines Been Established?

Research over the last 30 years has been employed to establish current air quality standards/guidelines to protect ecosystems. It should be noted that since 1985, the WHO has recognised that the protection of ecosystems against air pollution damage is relevant to human health, and appropriate guidelines have been established, which are currently equivalent to the UNECE's standards. The setting of both UNECE and WHO values have resulted from the outputs of workshops, where international experts have deliberated over the published work which provides the best information for this purpose.

The basis of our current knowledge, which is utilised in the setting of standards depends entirely on research carried out over the last 30 years, earlier studies having (in the case of SO_2 at least) indicated thresholds for injury of around an order of magnitude greater than is now known to be the case. In the case of O_3 two major research programmes have contributed to the setting of standards in the USA and Europe, respectively. In the USA the National Crop Loss Assessment Network (NCLAN) was established in the 1980s in order to derive dose response relationships between O_3 concentrations and yield loss in the ten most widely grown agricultural crop species. This involved experiments conducted to a highly standardised protocol in which plants were grown at five locations in open-top chambers, ventilated with charcoal-filtered air, ambient air or ambient air with different concentrations of O_3 added for 7 hours. The dose/response functions generated in this study were then combined with monitored and modelled O_3 spatial distributions to estimate the overall yield loss for each crop across the nation as a whole. The yield losses were then incorporated into an economic model, which indicated O_3-induced yield losses amounting to approximately \$3 x 10^9 per year.

Table 1 – UNECE Critical Levels/WHO Air Quality Guidelines

	O_3 AOT40 ppb.h	SO_2 μgm^{-3}	NO_x μgm^{-3}
Agricultural crops	3000	30	30 (all vegetation)
Natural vegetation	3000	20	
Forests	10,000	20 (15 in harsh climates)	
Lichens	-	10	

The UNECE critical levels for O_3 are based on a study across a range of European countries, with an essentially similar approach to NCLAN. In this case, spring wheat was employed as the experimental crop as it was believed to be highly sensitive to O_3, and again dose/response functions were generated. Unlike in the USA where the dose utilised was the 7 hour seasonal mean, the parameter employed is the AOT40. The latter is the accumulated exposure over a threshold concentration of 40ppb for daylight hours during the growing season. It takes into account experimental evidence that it is the peak concentrations of O_3 which are harmful, compared with means over different averaging times. Current critical levels, established by the UNECE and adopted by the WHO for its air quality guidelines are shown in Table 1. Air pollution control policy both internationally and within individual countries is aimed at reducing ambient concentrations to or below the critical levels in areas where particular receptors occur.

Critical loads were originally developed as standards to protect against the adverse impacts of acidification, particularly of lakes, being based on the buffering capacity of local soils. They are also employed to protect terrestrial ecosystems against acidification, using the soil calcium : aluminium ratio as the criterion. However, the role of acid deposition in damaging terrestrial ecosystems remains somewhat controversial, in contrast to aquatic systems. Thus the dire predictions of twenty years ago that acid deposition would destroy the forests of many parts of Europe by the present time have remained unfulfilled, although concerns remain about tree health and its relationship to air pollution.

However, there is much better evidence for excess nitrogen deposition causing widespread adverse effects on ecosystems in both the UK and elsewhere in Europe. Both oxidised and reduced forms of nitrogen have increased in their emissions massively over the last 50 years, due to the increase in the level of fossil fuel combustion (involving both stationary and mobile sources) and agricultural intensification (including both livestock production and inputs of artificial nitrogen fertilisers). In a wide range of nutrient-poor ecosystems, species adapted to conditions of low nutrient availability are being out-competed by species characteristic of more nutrient-rich environments. The former species may be stimulated in growth by increased nitrogen, but this can lead to an imbalance of nutrients and increased sensitivity to stresses such as drought, frost, infestation by insects and infection by fungal pathogens. Probably the best known example in this respect is the shift in lowland heaths from a heather dominated ecosystem to one in which coarse nitrophilous grasses are the major component. This problem is particularly severe in The Netherlands, where intensive livestock production has resulted in the highest nitrogen deposition in Europe. Changes in the growth, chemistry and phenology of heather renders it more susceptible to damage by frost and drought and attack by the heather beetle, with resulting die-back permitting the invasion of nitrophilous grasses. There are

Table 2 – Summary of proposed critical loads (kg-N $ha^{-1}y^{-1}$) for effects of nitrogen deposition on selected classes of vegetation

Ecosystem	Critical load	Effect of exceedance
Forests		
Acidic coniferous forests	7-20***	Changes in ground flora and mycorrhizae
Acidic deciduous forests	10-20**	Changes in ground flora and mycorrhizae
Calcareous forests	15-20 *	Changes in ground flora
Forests in humid climates	5-10 *	Decline in lichens
Heathlands		
Lowland dry heaths	15-20 ***	Transition from heather to grass
Lowland wet heaths	17-22 **	Transition from heather to grass
Upland Calluna heaths	10-20 *	Decline in heather and moss dominance
Arctic and alpine heaths	5-15 *	Decline in lichens, mosses and evergreen dwarf shrubs
Grasslands and wetlands		
Calcareous grasslands	15-35 **	Increase in tall grasses, altered diversity
Acid/neutral grasslands	20-30 *	Increase in tall grasses, altered diversity
Montane grasslands	10-15 *	Increase in tall grasses, altered diversity
Ombrotrophic bogs	5-10 **	Decline in typical mosses, increase in tall grasses

Quality of evidence supporting the proposed critical load:

*** reliable
** quite reliable
* uncertain

many other examples of undesirable changes in ecosystems arising from increased nitrogen deposition. These include the decline in *Racomitrium* moss at high altitudes in the UK and the failure of *Sphagnum* moss to recolonise the upland blanket bogs of the southern Pennines, where it was originally eliminated by high levels of SO_2, following the Industrial Revolution. The UNECE critical loads for nitrogen are largely based on field observations and in some cases are accompanied by a high degree uncertainty. They are shown in Table 2.

The Special Case of Lichens and Bryophytes

Lichens and bryophytes are known to be highly sensitive to SO_2, but less is known about their response to other pollutants. It has already been mentioned that *Sphagnum* moss was effectively eliminated from polluted areas of the southern Pennines in the early days of the Industrial Revolution. This resulted in degradation of the blanket bog ecosystem, with widespread erosion of the peat, which continues to the present day. Much more is known about the sensitivity of lichens than of bryophytes. Both groups of organisms are susceptible to atmospheric pollution because they do not possess the protective cuticle which is present on the leaves of higher plants and are adapted to take up their nutrients directly from the atmosphere. Since the Industrial Revolution the more sensitive species have been eliminated from large tracts of the UK countryside and until recently the larger cities and industrial areas were veritable lichen deserts. Indeed, the differential sensitivity of different lichen species to SO_2 has been exploited as a method of biomonitoring for this pollutant. As SO_2 levels have fallen there has been some reinvasion by lichens, but this is so slow that the biomonitoring scales utilised formerly have now broken down, with lichen distribution still to a large extent reflecting the SO_2 climate of some 30-40 years ago. However, one species of lichen – *Lecanora conizaeoides* – is showing the opposite response, with a decline in its abundance and distribution as SO_2 concentrations fall. This species was not recorded in the UK before the mid-19th century (although its inconspicuous appearance may have led to its being overlooked), and it has been suggested that it may have originated around sulphur springs in Iceland and that it may have a SO_2 requirement. An ongoing 23 year transect study by Imperial College into lichen abundance on oak trees from rural Sussex into central London has provided some evidence for the latter. At the start of this study, *L. conizaeoides* was present in very low abundance at the inner London suburb of Putney, the prevailing SO_2 concentrations of c.80 μgm^{-3} presumably being too high even for this species. As SO_2 levels fell by about 50% to the mid-1980s, there was a massive increase in the cover of this species at Putney. Then with a continuing fall in concentration it progressively declined, such that it now appears to be extinct at this location. This raises the possibility of *L. conizaeoides* being employed as a biomonitor of <u>falling</u> SO_2 concentrations. As SO_2 concentrations have fallen, nitrogen deposition has increased over much of Europe and nitrogen-loving lichen species and green epiphytic algae have likewise increased in many places.

Interactions with Other Stresses

It has already been mentioned that nitrogen deposition can predispose heather to other stresses, both biotic and abiotic. The same applies to NO_2, SO_2 and O_3. All three pollutants have been demonstrated at ambient concentrations to change the performance of both sucking and chewing herbivorous insect pests. All the evidence points towards these effects being mediated via pollutant induced changes in plant chemistry, which alter the host's nutritional value for the pest. In the case of SO_2 and NO_2 the response is nearly always to stimulate pest performance and this raises the possibility of large increases in the degree of infestation of vegetation in polluted areas, confirming many observations in the field in such places. For O_3 a more complex response is observed, with both negative and positive effects, which appear to have some dependency on environmental conditions.

There have been many examples of fungal pathogens showing increased or decreased incidence in polluted locations, including around industrial point sources, inside cities and alongside major roads. Two pathogens which have been studied in the field in the UK in relation to air pollution are blackspot of roses and tarspot of sycamore. Both species were formerly absent from areas with high SO_2 concentrations, but a recent study has shown unequivocally that they have invaded the urban areas where they were formerly absent. The exception is tarspot in London, where experimental studies have demonstrated that its spores are incapable of infecting sycamores within central areas of the city. Easily the most plausible explanation is that tarspot is susceptible to NO_2, with London having the highest concentrations in the UK of this pollutant.

The most studied responses to abiotic stresses are frost damage and drought stress on vegetation. There is abundant evidence for common phytotoxic air pollutants to predispose a wide range of species, including forest trees, native herbaceous vegetation and agricultural crops, to low temperature stress. Interactions with drought are complex, because air pollutants can cause increased opening or closure of the stomatal pores, which represent the route of entry of gases into the leaf, thereby changing the likelihood of injury taking place. Drought in itself will reduce stomatal opening and thus provide some degree of protection against pollution. On the other hand there is evidence for pre-treatment with air pollutants rendering plants more susceptible to subsequent drought stress.

It should be noted that some of these effects have been demonstrated to occur close to or below currently established critical levels. This raises the possibility that such indirect effects on vegetation may be equally or more important than the direct impacts addressed by these standards.

Future Developments

Present critical levels are a somewhat blunt instrument, based on limited information on a limited number of species. At present the UNECE is developing the so-called Level II approach. This is based on flux to the plant, rather

than atmospheric concentration. It takes into account prevailing climatic conditions which influence stomatal opening and, hence, flux of the pollutant into the plant. Thus models are predicting that with drier conditions prevailing in much of southern Europe, the higher O_3 concentrations there will produce lower yield losses than predicted on the basis of fumigation experiments with well-watered plants, due to a lower stomatal conductance. Of course, this situation will not prevail in the case of irrigated crops, which are likely to be particularly at risk in hot sunny climates with high levels of O_3. Very little is known about the response of native herbaceous species to O_3. Yet there is evidence that ambient O_3 levels can change species composition in mixed swards and induce selection for O_3 tolerance in some species, the latter confirming earlier work with SO_2. Thus a greater understanding of the response of a much wider range of species to O_3, using a flux-based approach will lead to the development of a more refined set of critical levels.

The evidence that plants can be harmed indirectly by air pollution at concentrations close to or below the critical levels, calls for much more research into this phenomenon, including derivation of dose-response functions. This could be used to refine further critical levels, incorporating an ecosystem-based approach. If the assumption is made in setting critical levels that the most vulnerable part of an ecosystem should be protected, then much more research is needed into impacts on lichens and bryophytes. While much is known about their response to SO_2, a much greater understanding is required of the impacts of O_3 and nitrogenous pollutants. Even in the case of SO_2 there may be issues to be resolved in that there is evidence that concentrations below the critical level of 10μgm^{-3}, set to protect cyanobacterial lichens, may have adverse effects on some of the more sensitive lichen species. It is well known that mixtures of pollutants can have synergistic interactions, in some cases producing substantially greater effects than the sum of the effects of the individual pollutants. Phytotoxic air pollutants rarely occur on their own in the field and yet critical levels are almost entirely based on studies on single pollutants. Clearly there is a requirement for research into the response of the more sensitive species to the ambient mixes of pollutants occurring in the field, with concomitant adjustment of air quality standards and guidelines.

Urban air quality is currently dominated by motor vehicle emissions. Thus it is surprising that so little research has been carried out into the direct effects of the mix of gases and particulates that come from vehicle exhausts. Current work at Imperial College and a number of other institutions, under the auspices of the Natural Environment Research Council's URGENT Programme is aimed at addressing this deficiency. A range of native plant species characteristic of urban areas are being exposed at different locations in the Natural History Museum's wildlife garden next to Cromwell Road and its kerbside monitoring station. A complementary series of experiments is being conducted in chambers at the Centre for Ecology & Hydrology, Bangor, in which the same species are fumigated with diesel exhaust gases and particles at roadside concentrations. This study is showing a range of responses to vehicle exhaust pollution, with stimulations and reductions in growth, changes in patterns of senescence and flowering, effects on gas exchange and leaf surfaces, and increased sensitivity to frost damage. Such effects demonstrate the need for further understanding of this topic, in particular whether planned reductions in vehicle emissions which are aimed at protecting human health, will also protect urban and roadside ecosystems.

Finally, all predictions of future impacts of air pollutants on vegetation should be viewed against the inexorable trends in rising CO_2 concentrations and its associated likely effects on climate. They should also be considered in relation to changing tropospheric O_3 characteristics. While peak O_3 concentrations in the developed world are falling as a result of controls in the emissions of its precursors, the background levels are continually rising in the northern hemisphere, probably largely due to increased emissions in developing countries. Predictions have been made that the mean background O_3 concentration could be as high as 70ppb by the end of the century. Such a situation is alarming in terms of likely impacts on vegetation. If ecological, forestry and agricultural systems are to be protected for the welfare of future generations then research needs to be generated to determine the likely combined impacts of O_3, CO_2 and climatic conditions on plant life. This is likely to lead to very different air quality standards and guidelines than at present.

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CONTENTS

NSCA produces information, organises conferences and training events, and campaigns on air pollution, noise and environmental protection issues. Founded in 1899, the Society's work on smoke control led to the Clean Air Acts. More recently NSCA has been influential in developing thinking on integrated pollution control, noise legislation, and air quality management.

NSCA's membership is largely made up of organisations with a direct involvement in environmental protection: industry, local authorities, universities and colleges, professional institutions, environmental consultancies and regulatory agencies. Individual membership is also available to environmental specialists within industry, local authorities, central government, technical, academic and institutional bodies.

Members benefit from joining a unique network of individuals who share an interest in a realistic approach to environmental protection policy; from access to up-to-date and relevant information; from reduced fees at NSCA conferences and training events. They contribute to NSCA's regional and national activities; to environmental policy development; to translating policy into practice; to the Society's wide-ranging educational programmes.

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Editorial

NEW TIMES ... NEW TECHNOLOGIES

From the end of this year, NSCA will begin publishing Clean Air and Environmental Protection exclusively as an on-line journal on our new website, which will be launched in the next few months. By removing the constraints of a printed journal, we can publish – more quickly and accessibly – articles of different lengths. This will include those that currently fall between what we would normally carry in Clean Air, and the very short pieces included in our monthly publication, NSCA Briefing. We will still publish those articles which are currently submitted, but they will now be available to a wider audience; previously published papers will also be available. It is planned to publish the on-line journal on a quarterly basis, with content updates signalled in NSCA Briefing.

The change in technology does not signal a change in the role and purposes of the journal. We still need a vehicle for articles which are accessible to specialists and non-specialists alike; and which are valuable contributions to information and debate rather than peer-reviewed technical papers for which the academic journals are the necessary medium.

Nor does the change signal a wider move from printed publications. With the principle purposes of Clean Air more effectively covered by on-line publication, there will be scope to explore other publication needs and opportunities. We may for instance want to explore the scope for annual reviews – in each of our separate subject areas such as air quality, noise, local environmental management and land quality – which take stock of key developments, briefly summarise current research issues and offer informed assessments of emerging issues.

The five papers carried in this issue cover a range of policy areas, but all cover issues about which there is currently much debate. The first, by Nick Hartley of OXERA Consulting, looks at the possible options for low carbon transport, something which will be vital for the UK in its future climate change policies. If we are to make anything close to the 60% reduction in carbon emissions for which the Prime Minister has signalled his support, we will need to take some fairly radical steps.

Noise from aircraft is an increasingly important topic, with the Government due to produce its aviation White Paper in the coming months. Much has been said on the environmental performance of the aviation sector, and the impact that expansion of our current infrastructure will have. The Parliamentary Office of Science and Technology (POST) has produced an extremely useful briefing note on aviation noise, which forms our second paper, to supplement its longer note on more general environmental impacts. Both can be found on the web at www.parliament.uk/post.

Our final three papers cover more technical areas. Derrick Crump, of BRE and one of our leading experts on indoor air quality, reports on a study investigating the migration of VOCs from contaminated land into buildings, which could have significant implications for the future use of such land. Two reports from Westlakes consultants, led by Vicky Auld and Richard Hill, look at two issues which have probably generated more discussion in the modelling community than any other – the validation of dispersion models and the impact of different Met datasets on dispersion model outputs.

REPORTS

Some Initial Thoughts on a Longer-Term Vision for Low Carbon Transport

Nick Hartley

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Previously Team Leader, PIU Energy Review

Introduction

This article looks beyond current concerns about low carbon car targets. It does not deal directly with the range of issues concerning fleet averages, "cutting edge", zero emission elements etc, though it may carry some implications for such concerns. It is primarily an attempt to place the low carbon transport agenda within the broader low carbon agenda.

For me, the fundamental question concerning "vision" is whether the low carbon agenda for transport should continue to be pursued as an entirely separate strand from the low carbon vision for the rest of the economy, and indeed for society.

The various studies which have been done of the carbon problem – notably the Royal Commission's report[1] – presented a very challenging picture. The achievement of 60% cuts in CO_2 emissions below year 2000 levels by 2050 will, it seems to me, not come about as a matter of natural progression in the economy. We know, for example from the PIU work,[2] that there is no prospect, on current trends, of meeting this target simply by changes in the power sector (the sector which has made the running in recent years as the UK has successfully achieved its first greenhouse gas targets). Even with a zero carbon electricity sector emissions would, on some projections, be well above the target level.

The long term role of carbon reductions in the transport sector is therefore apparent, and, of course, recognised, not least by the establishment of the Low Carbon Vehicle Partnership (LowCVP)[3].

Carbon Trading

The world as a whole is in a transitional state at the moment. In some countries the carbon agenda is clearly embedded in policy; in others less so. The last decade has seen a remarkable shift in perceptions. The publication of the recent UK Energy White Paper[4] marked an important step for the UK, when carbon policy, for the first time, took a central role in energy policy. Other countries have yet to reach this stage.

Much depends on the success of the next rounds of the international climate agreements. The pace of change may alter over time as this process meets various key hurdles. Yet the important point is that the carbon problem will, rightly, be addressed as a world problem. In this case, the central issue is how far carbon reductions should be made in the parts of the world where gains can be made most efficiently (i.e. cheaply) and how far reductions should be spread around to achieve global equity (and equity between "new" and "old" polluters).

One means of achieving global efficiency is, of course, to allow some form of carbon trading, via the Clean Development Mechanisms and Joint Implementation (i.e. schemes which allow rich nations to buy carbon reductions in poor ones). Even within a nation or group of nations, such as the EU, trading has the potential to produce unambiguous welfare gains, as sectors which find it easiest to make carbon emissions trade with those which face a harder task. These mechanisms are in their very earliest stages of development. There is a long way to go before we understand the full potential of trading (and its possible drawbacks).

However, the prospects for trading carry profound implications for long-term carbon policy, pursued nationally on a sectoral basis. Clearly any system of trading still needs sectoral regulation of carbon emissions, including systems of caps and measurement. But sectors are provided with much greater flexibility about how, when, and where, their mandated emission reductions might be achieved. I believe that if world and EU carbon trading schemes are found to be successful, there will be strong incentives to bring other sectors into the scheme. Indeed, the "general recital" covering the EU Parliament's amendments to the EU Directive on carbon trading already makes mention of transport (though this has no impact on the coverage of the scheme starting in 2005).[5]

The precise ways in which transport could be included within a trading scheme need substantially more thought. The most straightforward means would possibly be by means of an obligation on fuel suppliers to obtain carbon permits before they could sell fuel. But there are other possibilities.

It is clear that if transport fuel is included within a trading scheme, then the basis for transport fuel taxation would have to be reconsidered. But, in the UK, this is precisely what seems to be planned over the next decade if the Government does introduce widespread congestion charging. If we are heading towards a world where transport costs are much more precisely aligned to external costs, then some form of carbon taxation/charging would seem to be essential.

Encouraging Low Carbon Innovation

The issue is then whether a system of charging for carbon emissions could, or should, be the sole instrument used to pursue low carbon ends in the transport sector. Many would dispute that it could be. In particular, they believe that further instruments are needed to encourage innovation in the manufacture of cars and low carbon fuels.

In theory, if the world as a whole has a firm expectation that carbon constraints will remain in place for ever, and if the future world value of carbon had been established with reasonable certainty, then producers and consumers should be able to incorporate this knowledge in their consumption and investment decisions. In other words the long term carbon price would itself drive innovation. In practice, the future price of carbon may, for a long while, be in doubt, partly because the coverage of the world carbon trading system is still uncertain. But more important, many would dispute that consumers, in particular, are ever likely to be as rational or perspicacious as is implied in this view of the world. In these circumstances, there are limits to market-based responses, and there may be a case for government intervention to push innovation in "the right" direction.

The same case is made for intervention to favour renewable generation, as a means of helping the UK to meet its long-term carbon commitments. There would, for example, be no possibility of meeting current renewables targets if renewables were assisted solely by exemption from carbon trading rather than, as is the case, by the Renewables Obligation, a system of cross-subsidies which places a substantially higher value on carbon savings than the value coming out of present trading regimes.

The long-term issue is how, in the context of an international industry facing a global challenge, and, we may assume, a global set of obligations, innovation is best encouraged. The longer-term vision must, I believe, contain a commitment to ambitious new goals. But if global gains are required, it is by no means obvious that national commitments are the appropriate response, other than as an instrument of national industrial policy.

Encouraging Low Carbon Networks

Energy industries rely on networks. Thus an issue for energy policy is the extent to which certain desirable technological developments depend on the development of new networks to support their end-use.

- In electricity there is the prospect of each home having its own gas-powered Combined Heat and Power unit, producing both heat and electricity. This development will only be possible, at scale, if there are significant changes in the way local electricity distribution networks are run.
- Similarly, renewables generation in the far north of Scotland will only be possible if some means can be found of exporting the power to England.
- In the transport field, the potential for a hydrogen-based car transport system raises the prospect of huge new investment in the fuel distribution network, though the shape of this future network still seems, to me, to be very much in doubt. The different possibilities, as currently seen, seem to have very different costs.

The policy issue is how far the market can be relied upon to make these investment decisions. There seems little doubt that such investments require deep pockets in the companies concerned. There are such companies. But commercial investments of this kind will only be made if there is a stable, and encouraging regulatory framework, including within the planning system. The precise requirements of this framework need further thought in order for them to be part of a longer-term vision.

The transport community has a strong interest in the development of a low carbon power sector as a means of supporting the possible development of zero carbon production of hydrogen. I suggest that for the long-term vision the important point is not to take sides between different potential zero carbon electricity technologies – renewables or nuclear – but to note the growing integration between the various parts of the energy sector, as the low carbon agenda becomes ever more important for policy.

In future, as carbon targets tighten, there will be choices about where remaining carbon is best, i.e. most cost-effectively, emitted. It seems undeniable that these choices are best made within a market system, within which everybody receives good signals about the resource and environmental costs of the various alternatives.

Changing Public Perceptions

An essential part of a long-term vision should, I believe, be that transport policy is just one part of an economy-wide change whereby low carbon priorities are fully integrated in all aspects of the market place. There is clearly a huge distance still to be travelled, but if radical changes in carbon issues are to be achieved, then, it seems to me, essential that carbon considerations start to feature high on the list of consumers' priorities. Clearly they do not at the moment. This is a task not just for the transport industries, but for the government and all other sectors of the economy.

If we cannot reach the stage where consumers are themselves interested in the carbon impact of all their major purchases – and many would suggest that it is "pie-in-the sky" – then the task of making the economy-wide changes is likely to be more arduous than otherwise.

Conclusion

This article has sought to distinguish a number of different policy strands: each is important and each should be part of a long term vision. One element of the vision is that each should be clearly articulated and separated, so that the different intermediate objectives of policy – carbon savings today, innovation, the development of new networks, better consumer understanding – are clearly understood. The ultimate objective is, of course, a (very) low carbon economy.

References

[1] Royal Commission on Environmental Pollution (2000), *Energy – The Changing Climate.*

[2] Performance and Innovation Unit (2002), *The Energy Review.*

[3] More information on the Low CVP is available on www.lowcvp.org.uk.

[4] Department of Trade and Industry (2003), *Energy White Paper, Our energy future – creating a low carbon economy.*

[5] The initial coverage of the National Allocation Plan is energy activities (combustion installations with a rated thermal input exceeding 20MW; mineral oil refineries; coke ovens); production and processing of ferrous metals; mineral industry (cement; glass; ceramics; bricks etc); paper and board.

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Aircraft Noise

Parliamentary Office of Science and Technology

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For many living around airports, noise is the most evident environmental impact of aviation. This briefing examines the sources of noise from airports, the effects of noise on people and the implications of the Government's forecasts for the growth in aviation. Potential technical and policy options to reduce aircraft noise are outlined. A more detailed POST report is also available covering this and other environmental issues associated with aviation.[1]

Key Points:

- *current aircraft noise can affect the quality of life of half a million people living close to UK airports*
- *increases in air traffic could outstrip technological progress in making individual flights quieter and so worsen the noise climate around many of the UK's airports over the next 30 years*
- *as a result, more people could be affected by sleep disturbance, annoyance and possible health effects*
- *aircraft noise could constrain airport expansion unless substantial noise reductions are made.*

The Growth of Aviation

The Government has forecast that aviation is likely to grow over the next 30 years at an average rate of 4.25% per year.[2] These forecasts are based on unconstrained growth – i.e. that airport and airline capacity is provided to meet all demand. The Department for Transport (DfT) points out that previous mid-range forecasts underestimated demand, with actual demand either following, or even exceeding the 'high' forecast growth curve; particularly for forecasts made before the emergence of the low-cost airlines in the late 1990s.[3]

To meet the maximum forecast demand for flights, additional airport capacity would be required equivalent to that which can be handled through five new runways – with three of these located in SE England. Spare capacity exists at some airports (particularly at Stansted and Luton at off-peak times) and so some of the forecast growth could be met without the need for additional infrastructure. However, on current policies, spare capacity is likely to be exhausted by 2015, so unless demand were managed or congestion tolerated, meeting the forecast growth to 2030 would require new capacity.

Aircraft Noise

Sources

Aircraft landing and taking off are the chief sources of aviation noise. Individual aircraft have become quieter over the past 30 years, but flight frequencies have increased. As a result, aircraft noise is giving rise to increasing community concern. In particular, landing noise is increasing in importance, and has become the dominant reason for complaints at some airports. In addition, those living close to very large airports may experience 'ground noise' from sources on the airport such as taxiing aircraft, aircraft engine tests, generators or airside vehicular traffic. Transport links to an airport, particularly private vehicles and trains, can also make a significant contribution to noise around airports.

Box 1 outlines how aircraft noise is measured. In essence, the DfT estimates current and future impacts of aircraft noise by determining the area exposed to average sound levels of 57dB(A) or more between 7am and 11pm. This measure was chosen as an indicator of the onset of what the government describes as 'community annoyance' in the daytime, following a study in 1985 which showed a good correlation of this figure with annoyance. However, it is apparent that the mix and types of aircraft, their frequency of overflight, the social and economic circumstances of affected people and general levels of environmental awareness and sensitivity have changed since the early 1980s. The Government has therefore commissioned a three year study to provide a firmer basis for the relationship between aircraft noise and annoyance. The first results from this new study should be available towards the end of 2004.

Box 1: Measuring Sound and Noise

Environmental noise is measured with reference to the A-weighted decibel scale, dB(A). This reflects the fact that the human ear does not detect all frequencies of sound equally efficiently. To quantify sound levels which vary with time ***equivalent continuous sound level*** or $\boldsymbol{L_{eq}}$ is calculated. This indicates the average sound level over a particular time period. For example, an L_{eq}, 24h of 57dB(A) indicates that the sound energy produced by the noise source is equivalent to a constant sound of 57dB(A) over 24 hours. Other measures of noise are also available, that relate to different measurement periods, such as the instantaneous maximum noise level (L_{max}), or the average over certain periods, such as evening or night (L_{den}).

Aircraft Noise at Airports

The Government considers noise to have the potential for the onset of *significant community annoyance* above a level of 57dB(A) L_{eq}, but recognises that some people are annoyed at lower levels. Contours of noise from airports are drawn, showing the area exposed to average sound levels of 57dB(A) L_{eq} or more between 7am and 11pm. Contour areas are then compared with population data to determine the number of residents within that contour. Contours are calculated by summing and averaging the noise from arriving and departing aircraft. For example, at Heathrow in 2002, nearly 130 km^2 of land from Fulham to Windsor was within the 57dB(A) L_{eq} contour. Calculations of future noise exposure must also take account of the known or planned flight paths to and from the airport and (since different types of aircraft make different amounts of noise) the known or estimated fleet mix at that airport.

Box 2: The Effects of Aircraft Noise

Annoyance

Noise can lead to people feeling stressed and angry. It may interfere with conversations and leisure activities in the home, disrupt activities requiring concentration, and discourage people from using outdoor spaces. Further factors may affect whether noise is viewed as 'annoying':

- occurrence of exposure – noise may be more annoying if it occurs often, even if each noise event is quieter
- fear of accidents – concerns about air crashes may increase some people's sensitivity to aircraft noise
- fear of the future – especially about future growth in air travel and potential increases in the frequency of flights
- lack of control – inability to alter or escape from the noise source may make it more annoying.[6]

The subjective responses to aircraft noise makes it difficult to quantify the relationship between noise and annoyance. However, noise levels below 50dB(A) L_{eq} are unlikely to cause community annoyance while levels of 55dB(A) L_{eq} may severely annoy some people. In the UK, the DfT uses a level of 57dB(A) L_{eq} as an indicator of the onset of community annoyance in daytime. Nevertheless, there are likely to be people exposed to more than 57dB(A) L_{eq} who will not be affected, and also those exposed to lower levels who will be affected. The location of the 57dB(A) contour is therefore not a precise guide (see Box 3).

Sleep Disturbance

Interference with sleep patterns is frequently reported by those living near airports operating night flights. A recent study of residents in high noise areas close to Heathrow, Gatwick, East Midlands and Coventry airports found between 1 in 5 and 1 in 10 people often reporting difficulty getting to sleep or being woken early . The European Court of Human Rights has ruled that the UK Government's procedure for decision-making about night flights was flawed, and that this flaw amounted to a "*violation of the respect for private and family life and the home*" under the European Convention on Human Rights. This judgement did not state that night flights themselves were a violation of human rights. The Government is appealing the decision. In the meantime, night flights continue as before, but if the judgement is upheld, the Government would need to review the regulation and operation of night flights.

Other Effects

- ***cardiovascular effects*** – The WHO points to a 'weak link' between frequent exposure to loud noise and effects on the cardiovascular system, but has called for further research before it can offer any guidelines.
- ***mental health*** – there is limited evidence that noise can affect existing mental illness, but not cause it
- ***educational achievement*** – it is not clear whether noise affects school performance directly or from cumulative loss of teaching time where disrupted by loud noise.

Effects

The Government acknowledges that noise can be "*one of the most objectionable impacts of airport development.*"[4] Aircraft noise can affect concentration or sleep and result in feelings of anger, frustration and powerlessness to control the noise. These factors can thus adversely affect people's quality of life. However, while many express concerns over aircraft noise, there remain considerable uncertainties over the precise nature of its impacts. Box 2 outlines the key effects of aircraft noise. Overall, much of the research in this area is either contradictory or inconclusive and many, including the World Health Organisation have called for considerably more research. Evidence to date suggests that most people exposed to aircraft noise are not adversely affected, but more vulnerable groups may be at increased risk; particularly those with pre-existing sleep problems, stress or mental health problems.

Box 3: Airport Expansion: The Sydney Experience[7]

The opening of a third runway at Sydney's Kingsford-Smith airport in 1994 led to an immediate outcry from residents who found themselves significantly disturbed by noise, despite living outside the area designated as likely to be significantly affected during the planning process. This became a high profile political controversy, including creation of a single issue 'No Aircraft Noise Party', and led to the establishment of a Senate Select Committee on Aircraft Noise in Sydney. The Committee concluded that opening of the third runway had "*scarred a city*" and "*irretrievably complicated the future of airport development in Australia*", as well as being an "*environmental and social tragedy*". It also commented that the policy in Sydney at the time of concentrating noise pollution in one area was "a form of discrimination".

The Committee found that Sydney residents felt that they had been misled by use of noise contours to give an indication of likely noise impacts of the 3rd runway. Further, the Australian Department of Transport and Regional Services had proposed measuring noise exposure relative to the number of events above a given threshold. This implied that once noise reached a level high enough to be intrusive, the level of noise beyond this would be irrelevant. This relates to the relative importance of the frequency of noise events against the loudness of individual events in determining annoyance. Last, the case demonstrated that residents were most likely to be annoyed by and complain about aircraft noise if they felt they had been misled about it. The Committee found that providing user-friendly information about aircraft noise to prospective house buyers and tenants near major flight paths could have reduced complaints about aircraft noise.

Aircraft noise already has the potential to affect the quality of life of at least half a million people in the UK – with 80% of these living close to major airports in the southeast of England. Figure 1 shows the extent of noise pollution around five major airports in the UK under DfT's growth forecasts.[5] For each situation, under worst-case scenarios, more people are likely to be exposed. Under the central scenarios, increases will be expected at Manchester, Birmingham and Stansted. Reductions at Heathrow and Gatwick would result from technological improvements alongside severely constrained growth.

Managing Aircraft Noise

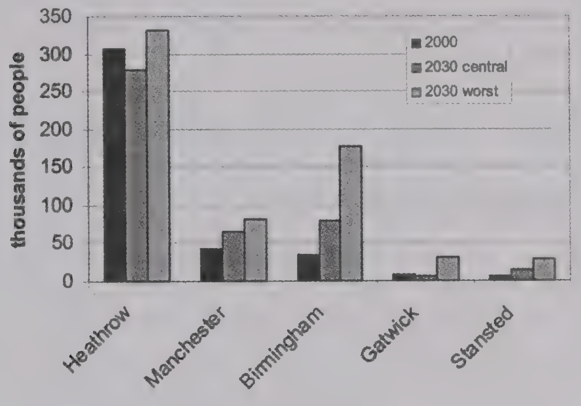
Aircraft noise in the UK is governed through international, EU and national regulation. At international level, the International Civil Aviation Organisation (ICAO) sets progressively tighter certification standards (known as Chapters) for noise emissions from civil aircraft. In addition to these specific requirements, the ICAO requires members to adopt a 'balanced approach' to airport noise management:

- reducing aircraft noise at source
- land-use planning
- changes to operational procedures
- restrictions on the use of the noisiest aircraft.

Figure 3

Forecast noise exposure[3]

(actual figures for 2000, central and worst-case forecasts for 2030)



Reducing Noise at Source

Over the past 30 years, improvements in aircraft technology have resulted in substantial reductions in the noise of individual aircraft. Around 20% of the current fleet already achieves a noise target 14dB better than the current (Chapter 3) standards. By 2004, British Airways expects 90% of its fleet to meet these levels of performance. Indeed, RollsRoyce reports that modern aircraft can achieve 18-24dB below the Chapter 3 standard. However, further improvements beyond the forthcoming Chapter 4 standards (which, from 2006, will require only a 10 dB(A) reduction on current standards) will be increasingly difficult to achieve.

Aircraft noise arises from engines and from the movement of turbulent air over the physical structure (airframe) of an aircraft. To date, noise reduction has focused mainly on reducing engine noise. This is now sufficiently low that tackling noise from the airframe is becoming as important (although it may be more challenging to reduce).

Technologies under development point to the perceived noise level from individual aircraft being halved, but translating laboratory-tested concepts into a fully functioning aircraft is difficult. In particular, the noise performance of a new aircraft is difficult to characterise fully before it is built and flown.

Land-Use Planning

Many UK airports are located in populated areas, so the potential for land-use planning to reduce noise exposure from existing airports is limited. However, planning has a role to play both for new developments near existing airports and for the development of new airports. The two main ways by which land-use planning can be used to help control aircraft noise are:

- planning permission – government planning guidance advises that planning permission for housing should normally be refused in areas exposed to noise from any source louder than 66dB(A) L_{eq} during the day (and 57dB(A) L_{eq} at night). At noise levels between 57 and 66dB(A) L_{eq} mitigation measures should be a condition on planning permission, but noise below 57dB(A) L_{eq} need not be considered.[8]

Box 4: Night-time Noise Control at Heathrow Airport

> There are restrictions on the total number of aircraft movements at night (11.30pm-6am) and the types of aircraft which can be used at night. In addition, there is a noise 'quota' for the total noise allowed at night at each airport over a whole season (summer or winter) – in effect, the noisiest aircraft are banned between 11pm and 7am. Aircraft used for night movements are assigned points according to how noisy they are, which count towards the noise quota. This can provide a powerful incentive to airlines to operate aircraft in the lowest categories possible for the size and type of aircraft, particularly for many long haul routes to the Far East, whose flights leave the UK in the late evening and arrive in the early morning.

- zoning – land around airports can be demarcated as either qualifying for compensation and support for noise insulation, or as being inappropriate for residential development given current or future noise levels.

Changes to Operational Procedures

The management of airspace for safety, navigation and logistical reasons leads to a concentration of air traffic along a small number of specific airways. The area on the ground affected by noise from departing aircraft depends both on the flight path followed, and on the rate of ascent of the aircraft. Controls on ***night noise*** at Heathrow airport are described in Box 4. There are three main ways to control ***take-off*** noise:

- noise preferential routes (NPRs) – where aircraft fly over the least populated areas after take-off
- managing thrust – maximum thrust generates extra noise close to the runway, but an aircraft gains height quickly. For residential areas, less thrust may reduce noise, despite the slower climb rate.
- concentrating or 'sharing' noise – an airport may adopt a policy to concentrate noise on a small number of residents under NPRs or to distribute it more widely.[9]

The Government acknowledges that where airports are close to populated areas, landing noise is increasingly a more serious problem. Here, final approach paths must operate in straight lines for safety reasons, so there is little flexibility in deciding which areas will be overflown. Reducing noise from landing aircraft has thus focused on the continuous descent approach (CDA). This is where an aircraft descends smoothly from around 6,000ft, usually at an angle of around 3 degrees, rather than descending through a series of level flights and steeper descents. This allows operation on low power and with low drag, minimising changes in engine tone.

A further option could be to adopt steeper descent angles so that aircraft are higher at any particular distance from the airport during their final approach. However, current air traffic control procedures would find it difficult to safely manage aircraft approaching at different angles. Were future aircraft designs and safety regulations to allow steeper approaches, it would remain challenging to use

these procedures while older and larger aircraft were using the same runway. It may be feasible however, to use longer runways for larger aircraft and shorter runways for smaller aircraft approaching more steeply.

Restrictions on the Use of the Noisiest Aircraft

Airports already impose restrictions on certain categories of aircraft at night (see Box 4). However, under an EU directive on aircraft noise (2002/30), restrictions on the noisiest aircraft can be introduced only after landuse controls and changes in procedures have been considered.

Further Measures to Reduce Aviation Noise

A number of further policy instruments could be used to reduce noise from aircraft, including:

- voluntary initiatives such as agreements between airports and local communities on the number or types of day and night flights; and between airports and airlines on procedures to minimise noise
- guidance to airport operators on potential mitigation measures such as a list of take-off noise limits realistically achievable by different aircraft types
- regulation such as legal sanctions for failure to follow noise preferential routes (track keeping), take-off noise limits, or the number of night flights, possibly enforced via fines on offending airlines or airports
- economic instruments such as landing charges that vary according to the noise performance of aircraft, or an airline's record on track keeping.

The Government is currently consulting on the use of economic instruments to tackle the environmental impacts of aviation. This is considered briefly in the previous POST report on aviation and the environment, but POST will also produce a specific briefing on this topic in the autumn of 2003.

The Limits of Technology

Historically, international regulation through the ICAO Chapters have been the main drivers for innovation to reduce engine noise. However, the forthcoming ICAO Chapter 4 standard is mandatory for new aircraft from 2006, and most new aircraft designs can easily meet these standards now. Some, such as the British Air Transport Association thus argue that these provide little incentive to go beyond current regulatory limits. Therefore, many now agree that national level controls (such as night noise quotas) are now stronger drivers towards innovation in engine and airframe technologies to meet environmental requirements.

However, while individual aircraft can be made quieter, the rate of innovation and uptake of new technology are likely to be much slower than the rate of growth of air travel. Consequently, within the Government's planning horizon for the future of aviation (to 2030), it is highly likely that following a period of relative improvement over the next decade, local environmental impacts from aviation could worsen. The question remains therefore over whether growth should be constrained to stay within acceptable limits, or whether the environmental impacts arising from meeting anticipated demand can be justified against other social and economic factors.

Endnotes

[1] *Aviation and the Environment*, POST report 195, Parliamentary Office of Science and Technology, April 2003. This report is available from POST. It is free to Parliamentarians, and costs £10 for non-parliamentarians. See below for details.

[2] *Air Traffic Forecasts for the UK 2000*, Department of the Environment, Transport and the Regions, 2000.

[3] *The future development of air transport in the United Kingdom: a national consultation,* Department for Transport, July 2002 (and second edition, February 2003).

[4] *Guidance to the Civil Aviation Authority on environmental objectives relating to the exercise of its air navigation functions*, Department for Transport 2002.

[5] This is related to noise above a threshold for 'community annoyance' adopted by the government. This threshold is 57dB(A) Leq and is discussed further in the full report.

[6] *Towards sustainable aviation,* Upham, P. et al (eds), Earthscan, 2003.

[7] *Falling on Deaf Ears*, Report of the Senate Select Committee on Aircraft Noise in Sydney, November 1995.

[8] *Planning Policy Guidance note 24: Planning and Noise,* Office of the Deputy Prime Minister, 1994 (and currently under review).

[9] The example of Sydney airport is apposite here. Previous policy was to concentrate noise but an Australian Senate inquiry described this policy as discriminatory and policy was changed to disperse aircraft noise over a wider area.

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VOCs in Buildings from Contaminated Land

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Volatile organic compounds (VOCs) occur in the indoor air as a result of emission from a wide range of sources and because they may be present in the outdoor air that enters the building through infiltration and planned ventilation. One possible source is ingress from the ground by diffusion through ground floor materials and the advective flow of air from the soil into the building. This can be of concern if the building is situated on or near contaminated ground.

There is no guidance available specific to the monitoring of VOCs in indoor air where ground contamination is of concern. This paper reviews available guidance on indoor air quality assessment and published studies involving contaminated sites to provide guidance on the measurement of VOCs in buildings that could be affected by ingress of contaminants from the ground.

Introduction

Volatile organic compounds (VOCs) occur in the indoor air as a result of emission from a wide range of sources and because they may be present in the outdoor air that enters the building through infiltration and planned ventilation. The indoor sources can be classified as those from building materials and furnishings, from the use of consumer products, release during combustion (of fuel and tobacco), and release from biological processes including from people and fungi. Many hundreds of compounds can be determined by available techniques and a number of studies in different countries have provided quantitative information for 80 compounds (Brown, 1999).

VOCs in indoor environments are of concern because of possible health effects on building occupants. These effects may be compound-specific and may be due to a mixture of compounds and will be dependent upon the amount and duration of exposure. People spend the large majority of their time indoors and indoor air is the major route of exposure to many air pollutants including VOCs (WHO, 1997; Crump et al, 1999; Mann et al, 2001).

External sources of pollution that may impact on the indoor air include VOC emissions from traffic and industrial processes. A further possible source is VOCs from the ground that diffuse through ground floor materials and are contained in the advective flow of air from the soil into the building. This can be of concern if the building is situated on or near contaminated ground. VOCs may then be present as a direct result of the contamination or, as in the case of landfill gas, they may be one of the products of decay of wastes.

It has been estimated that there are between 50,000 and 200,000 hectares of land affected by contamination in the UK (Environment Agency, 2001). Many sites will have been contaminated by industrial activities during the twentieth century and earlier, for which there may be few records. Not all brownfield sites will have contamination that poses a risk to health, but when considering re-development of such sites the possibility of contamination being present must be assessed. In 1998 the UK government set a policy target that 60% of new housing should be constructed on brownfield sites. This could involve some 2.2 million homes by 2021, based on predicted housing need. Therefore it is important that housing and other buildings are protected from ingress of organic vapours from contaminated land and that methods are available to evaluate risks and remediate problems.

This paper considers the methods available to study VOCs in indoor air that may originate from contaminated ground. It reviews available techniques and studies that have been applied elsewhere, and gives practical examples of monitoring in case studies.

The Role of VOC Measurement

VOC measurements are often undertaken as part of a standard suite of measurements to assess the indoor air quality of a building. When evaluating the results, the possibility of ground contamination should not be excluded, although other sources such as materials and occupant activities are usually of primary concern (Crump et al, 2002).

There are two main circumstances where VOC measurement in buildings may be undertaken specifically because of concerns about contaminated land:

- the investigation of existing buildings where a problem of indoor air quality due to contaminated land is suspected;
- measurements in new buildings to check the efficacy of remedial measures undertaken during development of a contaminated site.

For both of these cases there is a role for both modelling and measurement.

For the development of housing on brownfield sites in the UK, stepwise procedures have been developed to cover all activities ranging from initial enquiry to completion of remedial works (Environment Agency, 2001). There are four main phases:

i. risk assessment – hazard identification and assessment;

ii. risk assessment – risk estimation and evaluation;

iii. evaluation and selection of remedial measures;

iv. implementation of risk management.

These phases are further broken down into 11 steps and the last two of these include verification of remedial action and implementation of monitoring and maintenance programmes. The purpose of verification is to ensure that the completed works perform as required by the original risk management objectives. Examples of such verification activities are chemical tests of soil and groundwater, and monitoring of soil vapour concentrations and gas concentrations in under floor voids. The need for monitoring and maintenance will depend upon the risks and the type of remediation. It may be to ensure that the remedial works remain in good repair and it may involve monitoring, for example of groundwater and soil gas, over a period of months or years. The Environment Agency guidance does not provide any detailed guidance or refer to any other literature about verification and long term monitoring.

For the assessment of risk for new and existing buildings on sites that may be contaminated, the UK Environment Agency has published the Contaminated Land Exposure Assessment (CLEA) model for estimating child and adult exposures to soil contaminants for those potentially living, working and/or playing on contaminated sites over long time periods (Environment Agency, 2002a). The CLEA model has been used to derive soil guideline values appropriate for three types of land use for some contaminants. It is used to estimate average daily human exposure to soil contamination based on conceptual exposure models for each of the three defined standard land uses. Inhalation of contaminants in the indoor air is one exposure pathway included in the model. In this context, the CLEA model makes two assumptions:

i. that no free phase contamination is present (therefore the model considers only the equilibrium between contaminants sorbed onto the soil, in the soil solution, or as vapour);

ii. that the zone of contamination lies directly beneath the building (therefore transport through the ground is not considered).

Therefore, special consideration of the applicability of derived soil guideline values is required for sites where there are considerable quantities of free phase contamination and/or the contamination is at depths greater than a metre. The approach uses two different models to predict indoor concentrations, described as the Ferguson-Krylov-McGrath (FKM) model for buildings constructed with a ground bearing slab and the Krylov-Ferguson (KF) model for suspended floor construction.

A further report (Environment Agency, 2002b) reviewed and evaluated available soil vapour transport models to provide guidance on the choice of a suitable model for incorporation into the CLEA model. There are significant uncertainties with the models in many cases, and there is a need for a better understanding of the biological transformation of vapour-phase contamination, the role of advective transport in the soil subsurface and the collection and analysis of experimental and case study data on a variety of different housing designs. The review also included a comparison of four of the available models (GSI, BC, BPRISC and Ferguson) with actual field measurements concerning vapour transfer of petroleum hydrocarbons from the unsaturated zone into buildings. The main conclusions were that the predictions of indoor air concentrations from soil models should be considered as order of magnitude estimates. It also recommended that the BPRISC (Johnson and Ettinger sub-model) model should be used in preference to the Ferguson model for the CLEA framework.

Published Example Case Studies

During the planning stage of any study to measure organic contaminants in indoor air, it can be useful to take account of previous studies. BRE has reported results of investigations into complaints or concerns about poor indoor air quality in 263 buildings. Where causes were clearly established the large majority were found to be due to contaminants such as formaldehyde, naphthalene and white spirits from building materials (Brown et al, 1993; Brown et al, 1996). It is apparent that possible ground contaminants such as coal tar and solvents could potentially produce a similar mixture of VOCs in indoor air as the building material sources. Amongst the buildings investigated was one office building where the air was contaminated by VOCs characteristic of petrol vapour and this was found to be due to ingress into the building from contaminated ground. In a further case VOCs including norbornadiene were found to be entering the building from the ground.

The largest published UK study of indoor pollution due to contaminated land was undertaken by BRE and involved homes affected by the ingress of chlorinated butadienes (CBDs). The CBDs were a component of industrial waste that, prior to the 1960s, had been disposed of in disused quarries near Runcorn in North-West England (Crump et al, 2002a). Migration of the pollution resulted in contamination of the indoor air in properties adjacent to the quarries. The study was stratified in order to investigate temporal variation and the influence of building and environmental factors on the indoor concentration. It included development of a sampling and analytical procedure to determine the concentration of CBDs and the incorporation of extensive quality assurance and quality control procedures to ensure reliable information was available for a full risk assessment of the site (Wilkinson et al, 2002). The indoor investigation was designed as an integral part of a wider study to evaluate the amount and extent of the ground contamination and assess the risk to health of residents (McAlary et al, 2002).

As discussed by McAlary et al (2002), an assessment of a contaminated site should involve developing a conceptual model of the site. The aim is to establish whether there is a link between the contamination and a receptor that could be adversely affected by exposure to contaminants. Normally this initially involves a desk study of the history of the site and the known geology. It may then proceed by site investigations, including use of boreholes to further investigate geology and groundwater, and chemical analysis of soil and rocks and monitoring of vapour monitoring wells to characterise the ground contaminants. If the conceptual

model identifies a possible link between the contamination and inhalation exposure of building occupants then VOC measurements of the indoor air should be undertaken.

McAlary et al (2002) combined the use of geological investigation, vapour monitoring wells, building/pressure ventilation studies and indoor and outdoor air monitoring with the use of a model (Johnson and Ettinger) as part of the risk assessment process for the contaminated site near Runcorn. The alpha factor (defined as concentration of a vapour in indoor air divided by the concentration at a specified depth in the sub-surface) was calculated, based on field measurements, and simulated using the model. The alpha factors calculated using the model under-predicted the measured alpha factors by two orders of magnitude. However the discrepancy was considered to be due to site-specific conditions because the source of contaminants was to the side and not beneath the properties and the model is based on a linear concentration gradient from the source at depth to the building floor. When the source depth was assigned to be just beneath the building floor the predicted alpha factors were much closer to the measured values.

Krasnoff et al (2002) in the USA reported use of the Johnson-Ettinger model to predict exposure concentrations in a study of homes overlying groundwater contaminated with 1,1-dichloroethene. They comment that previous calibration and verification of the model had been limited to radon studies. Using model default parameters, they found the model under-estimated exposure concentrations up to 1000 times, though a much improved agreement between the model and measured indoor concentrations was obtained when site-specific values were used, based on soil gas and soil permeability testing.

The author is not aware of any published studies of post-development validation of the efficacy of remediation of a contaminated site involving monitoring of VOCs in indoor air.

Measurement Protocols

The author is not aware of any guidance specific for measuring organic contaminants in buildings that result from the ingress of soil vapour. However, very relevant background information is provided in a BRE Report (Crump et al, 2002b) that discusses a protocol for the assessment of indoor air quality generally. It provides guidance on designing a strategy for monitoring of indoor air quality to achieve different objectives including checking levels of particular pollutants against guideline values and investigating the cause of occupant complaints. It also provides information about the most appropriate methods of measurement including for VOCs and for methane and associated gases, including landfill gas.

An important consideration is whether the methodology that might be applied in a routine assessment of indoor air quality is appropriate for the study of a building influenced by contaminated land. Often the initial work to develop the conceptual model, possibly combined with industry profiles (Environment Agency, 2002c) that describe the types of contamination likely to be associated with particular historic land uses, can provide information about the type and order of magnitude concentration of VOCs of interest. However, the investigator should also be prepared for this information to be inadequate and, therefore, screening for a wider range of compounds (very volatile, volatile and semi-volatile) may be appropriate at an early stage of the work. The aim of this early assessment should be to define the compound(s) that are the main risk drivers for health effects due to exposure indoors and the study can then be directed to evaluating these in the most resource efficient manner.

Strategy

An important aspect of any investigation is the design of the sampling strategy and the selection of appropriate measurement methods. Crump et al (2002b) provide background information about sources of guidance for designing a strategy appropriate for particular objectives of an indoor investigation. They propose a strategy for assessing air quality in homes and office buildings in terms of acceptability for comfort and the risk to health. Also referred to are ISO standards under development that give guidance on monitoring indoor pollutants and a guide produced by a European expert group (ECA, 1994) on developing a strategy for sampling VOCs in indoor air.

There are also some other important considerations for designing a study of the impact of ground contamination including the following.

1. The study may not be concerned with specific defined buildings but an area on and/or adjacent to an area of contamination, either near the surface of the ground or at depth. Therefore the issue of the extent and the amount of the contamination is important for determining the area over which there is concern about ingress of contaminants into buildings. The area of concern may therefore include many buildings and these may be of different types (e.g. homes, offices, schools, industry). These will have a different pattern of occupancy and have occupants with varying sensitivity to the pollutants. Therefore defining the number of buildings that should be investigated, possibly as part of a phased sampling strategy can be an important initial step in the strategy.
2. The source will be the ground and this could influence the concentration in different rooms. This could be because the building may have more than one type of ground floor construction, a common occurrence where buildings have been extended. Different floors will offer varying resistance to ingress because of a range of factors including the type of structure (e.g. suspended or solid floor), the amount of cracking and gaps in the floor, the penetration of service pipes into the ground, and the presence of wall cavities. Also, properties with two or more storeys will have a greater volume than a bungalow and thereby dilute the pollutant in a well mixed property. However there can be a greater 'stack effect' in buildings with more than one storey and this can draw greater amounts of air and associated contaminants from the ground.
3. The amount of movement of contaminated vapour in the ground may be affected by the weather; for example soil permeability may be affected by rainfall, as will

groundwater levels, and changes in atmospheric pressure can influence soil gas movement. Therefore there could be temporal variation in the strength of the source due to weather conditions. It is also possible that the pollutant is migrating in the ground and the spread of influence may need to be monitored over a period of months or years if no active remediation is undertaken, or to evaluate remediation.

4. Occupant behaviour will influence the stack effect and ventilation of the property and this will also change with season. Therefore the concentration in the property would vary even if the source strength was constant.

The detail and accuracy with which the spatial and temporal variation needs to be defined depends to some extent on the nature of the health risk. If the risk is primarily due to long term exposure over a period of years, diurnal changes in concentrations may not be of concern for evaluating health risks. However, shorter term changes may be important if exposure to peaks in concentration is of concern. Primarily the investigator needs to ensure that the sampling strategy provides the appropriate information to enable an assessment to be made of exposure to the pollutant in order to assess the risk to health.

Sampling and Analytical Methods

The strategy should define the list of target compounds to be monitored, the sensitivity of the method and the sampling duration. Other considerations should include the required performance of the method in terms of uncertainty of measurement. It may be necessary to modify the strategy because of limitations with the capabilities of available methods and the resources available for the investigation.

Crump et al (2002b) describe the two main approaches for sampling VOCs in air, which are pumped sampling (normally for a period of 30 minutes to one day) and diffusive sampling for a period of days or weeks. Guidance on the measurement of VOCs in workplace, ambient and indoor air is available in ISO EN standards 16017-1 and 16017-2. There are also more specific standards for monitoring carbonyl compounds in indoor air (ISO 16000-3 and ISO 16000-4) and an ISO standard (16000-6) is under development for measuring VOCs (C_6-C_{16}) in indoor air using pumped sampling. These standards provide guidance, but may not provide a method suitable for assessing a particular site and therefore the investigator should be prepared to develop and adapt methods for particular investigations. In particular if the study involves the need to determine very volatile and semi-volatile compounds that are outside the range of compounds covered by ISO16000-6 (C_6-C_{18}), then considerable work may be required to develop a validated method based on the guidance in ISO16017-1 and 16017-2.

Conclusion

Contaminated land is one possible source of VOCs in buildings and should be a consideration during indoor air quality investigations. When studying sites with known or suspected contamination, measurement of VOCs in indoor air may be required for evaluating the risk to health posed by the contamination and for validating the performance of remediation measures undertaken to control any risk.

There is no guidance available that is specific to the monitoring of VOCs in indoor air where ground contamination is of concern. There is guidance on the approach to assessing the risks from a contaminated site and on assessing the effectiveness of remediation. A number of published documents provide useful background for the study of buildings on or adjacent to contaminated land, for example the protocol published by BRE for general indoor air quality assessment, European and international standards concerning strategies and sampling, and analytical methods for the determination of VOCs in air. The strategy, including the selection of methods, needs to take account of the type and extent of ground contamination, the pathways of ingress into the building, the building type and pattern of exposure of occupants and the nature of the health risk. Also to be considered is the possible temporal and spatial variation in concentrations that may occur as a result of contaminant migration, and the influence of climate as well as building and occupant characteristics on the ingress and dilution of VOCs in the indoor air.

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Configuration and Validation of Regulatory Atmospheric Dispersion Models for a Complex Industrial Site

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Gaussian plume atmospheric dispersion models are widely used for assessing the environmental impacts of industrial sites in the UK. The regulators of these sites require that models are fit for purpose, have been validated and have been independently reviewed. This paper discusses some of the aspects that should be considered when configuring Gaussian plume models for a complex industrial site, using the BNFL Sellafield site in West Cumbria as a case study. For this particular site a major consideration was the effects of the multitude of site buildings on dispersion and the incorporation of the detailed on-site meteorological data.

The emission of the radioactive noble gas krypton-85 from this site allowed the independent validation of the "next generation" Gaussian plume models ADMS and AERMOD and the older NRPB R91 atmospheric dispersion model. The results showed that no particular model consistently outperformed all the others. However, similar performances were found from AERMOD and ADMS when the site buildings were included using their respective buildings effects modules, with 32% of the predictions of both models being within a factor of two of the field measurements. Furthermore, the NRPB R91 model provided similar results to the AERMOD and ADMS models, when used with effective stack heights to consider building effects, demonstrating that this model continues to be a useful tool.

Introduction

The Environment Agency's policy statement on the choice of air dispersion models, EAS/2007/1/1, (EA, 2000) clearly states that the Agency does not prescribe or favour any particular model, though the EA does require that the model is fit for purpose, has been validated and is well documented. This policy is reflected in a similar statement produced by the Royal Meteorological Society (RMS, 1995) which expands upon the issues of justification of the choice of models by noting that detailed site-specific considerations should be made to the types of features required in a modelling assessment including, in its examples, such influences as coastal effects, terrain and buildings. RMS (1995) also expands on the issue of validation noting that model users should give an account of the range of conditions for which the model has been validated and comment on how appropriate these may be for the type of assessment they are conducting. These policy statements leave a considerable scope for operators to decide upon which model should be used for a particular assessment. However, decisions related to the fit-for-purpose of a particular model, or whether it has been adequately validated, for what may be site-specific conditions, are difficult to qualify.

The majority of regulatory assessments of industrial discharges to the atmosphere, both within the UK and overseas, use Gaussian plume models. These models benefit from being relatively simple to set-up, can be efficiently run on standard desktop PCs and, for many cases, the progression to more detailed modelling methods (such as computational fluid dynamics) is difficult to justify when one considers the uncertainties inherent in these assessments due to uncertainties in model inputs (principally meteorological and emissions data) and from the natural variability in the turbulent atmosphere. The most commonly used Gaussian plume models in the UK fall into two distinct groups that are loosely based on the age of the underlying dispersion theories.

First Generation Gaussian Models

Simple Gaussian plume models, such as the NRPB R91 model (Clarke, 1979) or the US-EPA ISC model (EPA, 1995) use the semi-empirical curves of Pasquill and Gifford (discussed in Gifford, 1976) and Briggs (1974) to parameterise lateral and vertical rates of dispersion. The variation in the rates of plume dispersion due to thermal effects on atmospheric turbulence is included in these models by subdividing the atmosphere into discreet stability classes.

The main advantage of using a First Generation Gaussian plume model is the simplicity of the method, allowing dispersion calculations to be undertaken extremely quickly using a computer, or even by hand. This enables a degree of transparency in the calculations, which is not possible with more computationally expensive modelling methods, and has strongly influenced the decisions of both national and international regulators of nuclear licensed sites to continue to use these models.

The simplicity of this modelling approach is also its major drawback, as these models do not allow dispersion rates to vary with height in the atmosphere and, as such, are only strictly applicable to the source heights used for deriving the original dispersion curves, which in the case of the Pasquill-Gifford method, restricts their application to near surface releases. This is interpreted in NRPB R91 (Clarke,1979) as referring to sources with effective release heights of up to 200 m. Further limitations arise due to the use of a discrete stability classification system to describe the continuous variations that occur in actual meteorological conditions, creating unrealistic step changes in plume behaviour at the transition points between classes. Furthermore, many tracer dispersion experiments have

shown that vertical concentration distributions can only be regarded as being Gaussian during the most stable conditions and are in fact typically closer to being exponential in shape (Pasquill and Smith, 1983).

Gaussian plume models principally treat dispersion from isolated sources in extensive flat terrain and so need to include some modifications to treat the complex features that characterise many industrial sites. Jones (1983) describes some simple modifications that can be made to the NRPB R91 model to account for buildings, plume rise and coastal effects. These modifications involve using:

- Virtual source models to account for the entrainment of near-roof-top releases into the cavity wake of a building or to account for the reduced rate of dispersion as a plume moves through the stable maritime boundary layer.
- Effective stack heights to model the rise of a buoyant plume, or the fall of a plume affected by building downwash.

New Generation (Quasi) Gaussian Models

Advancements in Gaussian plume atmospheric dispersion models have focused on improving model predictions by moving away from the system of discrete stability classes toward more realistic descriptions of boundary layer meteorology, based on the Monin-Obukhov length scale (termed L_{MO}) and the height of the boundary layer. These methods allow new generation models, such as AERMOD (Cimorelli et al, 2002) and ADMS (Carruthers et al, 1994) to account for the vertical variation in atmospheric turbulence and so remain valid for the full range of release heights. They also allow ADMS and AERMOD to realistically treat the interaction of a dispersing plume with the inversion layer that caps the boundary layer, and, the use of a continuous scale in these models removes the step changes in behaviour that result from using a stability classification scheme. A further advancement in the underlying descriptions of atmospheric dispersion in AERMOD and ADMS is the treatment of the non-Gaussian shape of the vertical plume dimensions in unstable meteorological conditions.

ADMS and AERMOD both can treat some of the effects of buildings and terrain on dispersion. ADMS uses the FLOWSTAR model to resolve flow and turbulence across valleys and hills and also to account for changes in surface roughness due to variations in land use, though it should be noted that FLOWSTAR is only recommended to be used when terrain gradients are between a lower limit of 1:10 and an upper limit of between 1:3 - 1:2 (Carruthers et al, 2000). AERMOD contains a much simpler model of terrain effects than ADMS that evaluates how a plume interacts with a hill on a sliding scale between two behaviours: (i) where the plume impacts upon the hill; or (ii) where the plume follows the terrain contours over the hill.

The effects of site buildings on dispersion are modelled in ADMS and the latest versions of AERMOD (02222) and ISCST3 using relatively simple schemes developed from the results of a large number of wind tunnel studies. It should be noted that an earlier version of AERMOD (99211) used the much simpler BPIP buildings module which applied some of the same considerations as used in Jones (1983) with the NRPB R91 model. These buildings models treat the entrainment of material into the wake cavity of a building and also model the effects of streamline deflections on an elevated plume. Their principle limitation is that both the ADMS building model (ADMS-BUILD) and the corresponding US PRIME model can only determine the effects of a single building, and consider groups of buildings by sorting them into a single "effective" building. One of the major differences in the treatment of buildings between AERMOD and ADMS is the method used to determine the appropriate "single effective building". AERMOD allows the user to input up to 75 site buildings and then applies the US Good Engineering Practice stack height and building downwash guidance to determine the parameters of the single effective building on 36 wind sectors surrounding each emission source. The procedure in ADMS is somewhat different in that fewer site buildings can be input (10) though the user is provided with more scope in determining which is the dominant building for each emission source and the program calculates the dimensions of the single effective building for each source for every line of meteorological data.

The influence of stack parameters such as the temperature and velocity of the release are included in ADMS and AERMOD to determine plume rise. ADMS also allows modellers to consider a wide range of complex effects that are not currently included in AERMOD, suggesting that it is applicable to a wider range of sites. It should be noted that there are limitations in which of these options can be combined, or used in conjunction with the Buildings and Terrain options. The additional complex effects ADMS can model are:

- Gas and particle wet and dry deposition.
- Simple NO_x chemistry.
- Coastal effects caused by the change in boundary layer structure as a plume disperses inland.
- Radioactive decay and external doses caused by immersion in a radioactive plume.
- Concentration fluctuations and dispersion of short-term puff releases.

A Case Study on Model Configuration and Validation - BNFL Sellafield

The above section has highlighted the similarities and differences between the commonly used modelling methodologies that are applied to assess industrial discharges in the UK, demonstrating that the selection and justification of a particular model is often not straightforward, particularly for sites where it may be argued that a range of interacting complex effects should be considered.

Features of the site that might influence dispersion modelling assessments

A good example of a complex site is BNFL Sellafield in West Cumbria, a location that is principally involved in the reprocessing of spent nuclear fuel and development of future nuclear technologies. This site extends over an area of approximately 2.5 km^2 and presents a number of challenges for dispersion modellers, these include:

- Multiple discharges from stacks ranging in height from 8 m to 125 m.
- Effects of multiple site buildings on dispersion, with buildings ranging to a height of more than 60m.
- Coastal effects.
- Terrain effects due to both variations in terrain height and land use.
- Integration of the detailed meteorological data recorded at the Sellafield meteorological tower in assessments.

Configurations of the three main regulatory atmospheric dispersion models (NRPB R91, ADMS and AERMOD) that could be applied to the Sellafield site have been developed in order to capture the complex site effects discussed above.

• *NRPB R91 Configurations*

The NRPB R91 model is currently widely used throughout the nuclear industry for reasons of transparency, speed and simplicity and is regularly configured in two modes, either to calculate air concentrations at specific crosswind positions (termed off-axis) or to calculate air concentrations averaged laterally across a 30° wind sector (termed sector averaged). A slight modification to the NRPB R91 code was also tested whereby wind speeds were extrapolated to the stack height for off-axis calculations. The effects of the site buildings were included in this model using the effective stack height method (Fulker and Singh, 1994), where model predictions were compared with wind tunnel measurements of air concentrations to determine the appropriate stack height to account for plume downwash. An example graph showing an effective stack height calculation is shown in Figure 1.

The standard meteorological data used in the NRPB R91 model is wind speed, wind direction and an estimate of the stability class. Gradients of wind speed and temperature that were measured at the Sellafield meteorological tower were evaluated to determine Monin-Obukhov stability lengths using flux profile methods (as discussed in Monteith and Unsworth, 1990). Stability classes were determined using the relationship between stability length and stability class determined by Golder (1972). A further consideration that was made was to include the effects of wind meandering using the default semi-empirical relationship contained in NRPB R91 and also through inputting the measured values of the standard deviation of the wind direction (termed σ_θ). Overall three NRPB R91 configurations were identified, these are shown in Table 1.

Table 1: Definitions of the 14 different model configurations used for the Sellafield site

No.	Model	Building Effects	Terrain Roughness	Terrain Height	Coastal Effects	Met Data
1	NRPB	ESH^{ab}	0.3 m	Flat	No	PG
2	R91	ESH^{c}	0.3 m	Flat	No	PG
3		ESH^{b}	0.3 m	Flat	No	PG + σ_θ
4	ADMS	BUILD	Map	Flat	No	FP
5	3.0	BUILD	Map	Flat	No	FP + σ_θ
6		BUILD	Map	Map	No	FP + σ_θ
7		ESH	0.3 m	Flat	No	FP
8		ESH	0.3 m	Flat	No	FP + σ_θ
9		ESH	0.3 m	Flat	Yes	FP
10		ESH	0.3 m	Flat	Yes	FP + σ_θ
11	AERMOD	BPIP	0.3 m	Flat	No	AERMET
12	99351	ESH	0.3 m	Flat	No	AERMET
13	AERMOD	PRIME	0.3 m	Flat	No	AERMET
14	02222	ESH	0.3 m	Flat	No	AERMET

DEFINITIONS: ESH: Effective stack height; PG: Stability estimated using Pasquill-Gifford classes; σ_θ : Standard deviation of the wind direction; FP: Stability estimated using flux profile techniques.

NOTES: a: lateral concentrations assumed to be constant across a 30° sector; b: wind speed used in the dispersion calculations extrapolated from 10 m to the stack height; c: wind speed used in the dispersion calculations fixed at 10 m for all stack heights (default on-axis configuration from Clarke, 1979)

• *UK ADMS Configurations*

One of the main difficulties in applying the ADMS model to such a complex site as BNFL Sellafield is in deciding how to treat the large number of site buildings. Inputting all the buildings on the site was not possible and initial attempts to manually combine buildings into 10 building groups resulted in unrealistic output from ADMS-BUILD. A more robust approach was identified whereby the closest building to each stack was specifically modelled using the ADMS-BUILD module, whilst the influences of the surrounding buildings on the site were included by defining a roughness length map and running FLOWSTAR to determine their effects on dispersion. As the immediate area surrounding BNFL Sellafield is relatively flat, with only a few points where terrain gradients exceed 1:10 (the cut-off point for applying FLOWSTAR), it was unlikely that terrain height would significantly influence dispersion. However, in order to establish the validity of this assumption a model configuration was created that included buildings, terrain roughness and terrain height.

Effective stack heights were also determined for use in ADMS from wind tunnel data, in order to provide a comparison with the predictions of the ADMS-BUILD module. A further advantage to the effective stack method was that its use enabled the consideration of the relative importance of coastal effects, which cannot otherwise be modelled by ADMS in conjunction with buildings and terrain. The application of the ADMS Coastlines module required data on sea temperatures that were determined, on

a monthly time resolution, from measurements collected by Kennington et al (1997-1999).

Meteorological data were input into ADMS as wind speed, wind direction, near surface air temperature and Monin-Obukhov stability length. As with the NRPB R91 model the effect of including σ_θ values in the meteorological input file was evaluated, resulting in six different ADMS configurations, shown in Table 1.

• AERMOD Configurations

AERMOD uses the AERMET preprocessor to derive boundary layer parameters, such as the Monin-Obukhov length and inversion height, from routine meteorological data collected by the US National Weather Service (US-NWS). These data all include some estimate of the cloud cover, which is required by the pre-processor to determine the stability of the atmosphere. As cloud cover observations are not collected at the Sellafield meteorological station it was not possible to run AERMET directly with these data. Instead, cloud cover data were calculated using their relationship with solar radiation, as discussed in Thomson (2000). Data on the Julian day, hour of day, surface temperature, wind speed and direction (at 10 m), precipitation and the calculated cloud cover were compiled into ADMS formatted meteorological files which were then run through the ADMS to SAMSON (a US-NWS format) converter to derive an AERMET (AERMOD's meteorological pre-processor) input file. This file was then processed to derive AERMOD input in combination with a file containing the measurements of wind speed, wind direction and air temperature at each of the eight heights on the 48 m tall meteorological tower at Sellafield at a 10 minute time resolution.

The processing of meteorological data for input into AERMOD was considerably more complex than required for ADMS and involved writing computer code to ensure that formatting conventions were observed. It should be noted however that for the majority of regulated sites these problems could be overcome by simply buying data in the correct format and are only likely to be encountered when dealing with similar industries that have invested in their own meteorological stations. Operators of major installations need to balance the advantages, or otherwise, of taking on-site meteorological measurements or relying on remote monitoring stations, particularly when these stations are decreasing in number.

The configuration of AERMOD to include the effects of buildings on the Sellafield site was relatively straightforward as the model can consider input of up to 75 buildings, hence all the major structures on the site were able to be entered. The effects of buildings on dispersion were modelled using the latest version of AERMOD (02222), which incorporates the PRIME buildings module, and using the previous version (99211), which applies the simpler BPIP routines. In order to provide a comparison with the results from NRPB R91 and ADMS, effective stack heights were also derived for use in AERMOD. Overall four configurations of AERMOD were created, as listed in Table 1.

Validation of the Model Configurations

The discharge of the weak beta emitting radioactive noble gas, krypton-85 (^{85}Kr) from the nuclear fuel reprocessing operations on the Sellafield site provides a useful tracer for validation experiments. These releases occur from two stacks (THORP and MAGNOX, which are 125 m and 122 m tall respectively) and emission rates are accurately measured using in-stack monitors. Field experiments have been conducted since 1996 to determine off-site ^{85}Kr air concentrations and measurements have shown that upwind background ^{85}Kr air concentrations are low (approximately 1.4 Bq m^{-3}) as Sellafield is one of the few significant sources of this radioisotope worldwide.

Measurements of ^{85}Kr air concentrations were made by Westlakes Scientific Consulting using a cryogenic distillation technique (see Janssen et al, 1986) resulting in the collection of 211 samples. In order to assess local dispersion around the site, and thus provide a validation of the setup and assumptions used in modelling the site buildings, measurements were only used in the validation study when they had been collected within 3 km of the site. Details of these measurements are shown in Table 2.

Table 2: Details of the field measurements of ^{85}Kr air concentrations used in the model validation study

Duration	Number of measurements	Air concentration (Bq m^{-3})		
		Maximum	Minimum	Average
2 hours	55	5510	3.2	513
15-48 hours	120	1543	1.2	136
164 – 168 hours	13	364	1.9	55
All	188	5510	1.2	240

A model validation database was constructed to contain all the data collected during the field experiments, a screenshot from which is shown as Figure 2. This database also contained computer code to:

- Automate the running of NRPB R91, AERMOD and ADMS.
- Import the simulation results.
- Conduct statistical analyses of selected data periods.

The statistical analyses evaluated the mean bias of model predictions, the fraction of predictions within factors of 2, 5 and 10 of the field measurements, the correlation coefficient (R^2) and an expression of the scatter between modelled and measured data through the Normalised Mean Squared Error (NMSE). The mean bias statistic was defined as the overall average of the model predictions divided by the corresponding average of the field measurements. The remaining statistics are a common suite of analyses often used in model validation studies.

The results of the model validation study are shown as a scatter graph in Figure 3. The results demonstrate that the models tended to be more accurate at predicting high concentrations with a marked drop in performance at concentrations below 100 Bq m^{-3}.

Figure 3 shows clearly the scatter in results from the model configurations; however in order to quantify the performance of the models a statistical analysis of these results was conducted. Table 3 shows results of the statistical analysis, demonstrating that similar features were observed when AERMOD and ADMS were run using their buildings modules (configs. 4, 5, 6, 11, 13), with 32% of the model predictions being within a factor of two of the field measurements respectively and very low mean bias statistics being recorded. The predictions of ADMS, AERMOD and NRPB R91 using the effective stack height method also showed some common features, with all these configurations showing a mean bias substantially greater than 1 and thus providing conservative estimates of long term average air concentrations. The effective stack height method also provided the highest F2 scores overall with NRPB R91 (config. 3) and ADMS (configs. 7 & 9) being within a factor of 2 of the measured air concentration for 35% of the time.

Table 3: Results of the statistical analysis of model performance

Model	Config.	MB	F2	F5	F10	R2	NMSE
NRPB R91	1	2.26	0.27	0.51	0.69	0.15	14.39
	2	4.20	0.20	0.53	0.68	0.32	22.74
	3	2.32	0.35	0.59	0.75	0.36	9.19
ADMS	4	1.16	0.29	0.60	0.78	0.33	5.21
	5	1.11	0.32	0.67	0.80	0.37	4.68
	6	1.12	0.31	0.64	0.79	0.34	5.11
	7	2.85	0.35	0.61	0.77	0.50	18.92
	8	2.80	0.31	0.59	0.77	0.52	17.55
	9	2.76	0.35	0.62	0.77	0.49	19.34
	10	2.71	0.32	0.60	0.77	0.51	17.93
AERMOD	11	1.35	0.30	0.61	0.74	0.40	3.99
	12	2.75	0.20	0.48	0.69	0.61	13.11
	13	1.07	0.32	0.65	0.78	0.41	4.06
	14	2.52	0.27	0.54	0.74	0.61	10.01

DEFINITIONS: MB: Mean Bias; F2, F5, F10: Fraction of model predictions within factors of 2, 5 and 10 of the field measurements; R2: Correlation statistic squared; NMSE: Normalised Mean Squared Error.

The statistical analysis demonstrates that no single model configuration outperformed all the others for all of the tests. However, it is useful to estimate the overall relative performance of the model configurations, hence ranked scores were calculated, where each configuration was ranked from 1-14 (14 being the best performing) for each test and the overall scores were summed. The resulting ranked scores are shown in Figure 3. These show that overall the ADMS and AERMOD models performed the best, when configured using their buildings modules. The remaining configurations of ADMS and AERMOD (including one of the ADMS buildings configurations, 4) and configuration 3 of NRPB R91 were all found to perform similarly with the two simplest NRPB R91 configurations (1 and 2) performing the poorest overall. Interestingly, the inclusion of values of the standard deviation of the wind direction in the meteorological input files made a considerable improvement to the ADMS-BUILD and NRPB R91 configurations; though little improvement was found when coastal effects or terrain heights were modelled with ADMS.

Conclusions

The choice and justification of which Gaussian Plume model to use to determine dispersion from a complex industrial site is often not straightforward. Modellers have to attempt to identify the most relevant aspects of dispersion which may need to be modelled and determine the extent (and likely realism) to which these can be modelled with existing software. A case study using BNFL Sellafield was presented, highlighting some of the difficulties of modelling such a complex site and deriving a number of different model configurations that addressed these problems. Overall the ADMS and AERMOD models, when configured using their buildings modules, showed the strongest performance, though the NRPB R91 model provided comparable results to the next generation models when configured using effective stack heights. This demonstrates that NRPB R91 continues to remain fit-for-purpose for conducting conservative and transparent radiological assessments.

This study highlighted that modelling coastal effects and terrain heights did not significantly improve the realism of the model predictions for this site, though including the standard deviation of the wind direction in the meteorological input to ADMS-BUILD and NRPB R91 seemed to improve predictions overall. Clearly, the choice of model is only one of the uncertainties that is encountered when modelling a complex site and often may not be the dominant one.

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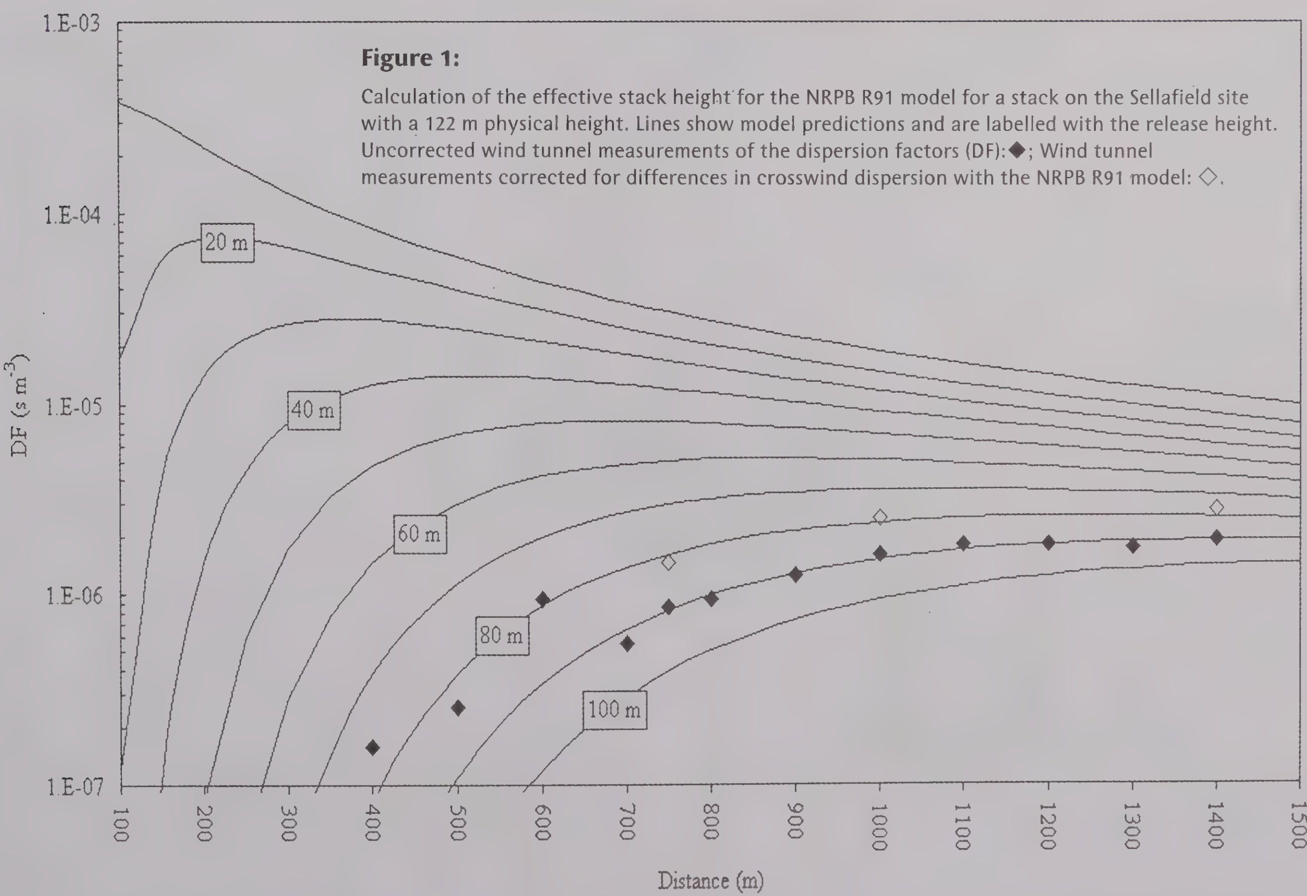

Figure 1:

Calculation of the effective stack height for the NRPB R91 model for a stack on the Sellafield site with a 122 m physical height. Lines show model predictions and are labelled with the release height. Uncorrected wind tunnel measurements of the dispersion factors (DF):◆; Wind tunnel measurements corrected for differences in crosswind dispersion with the NRPB R91 model: ◇.

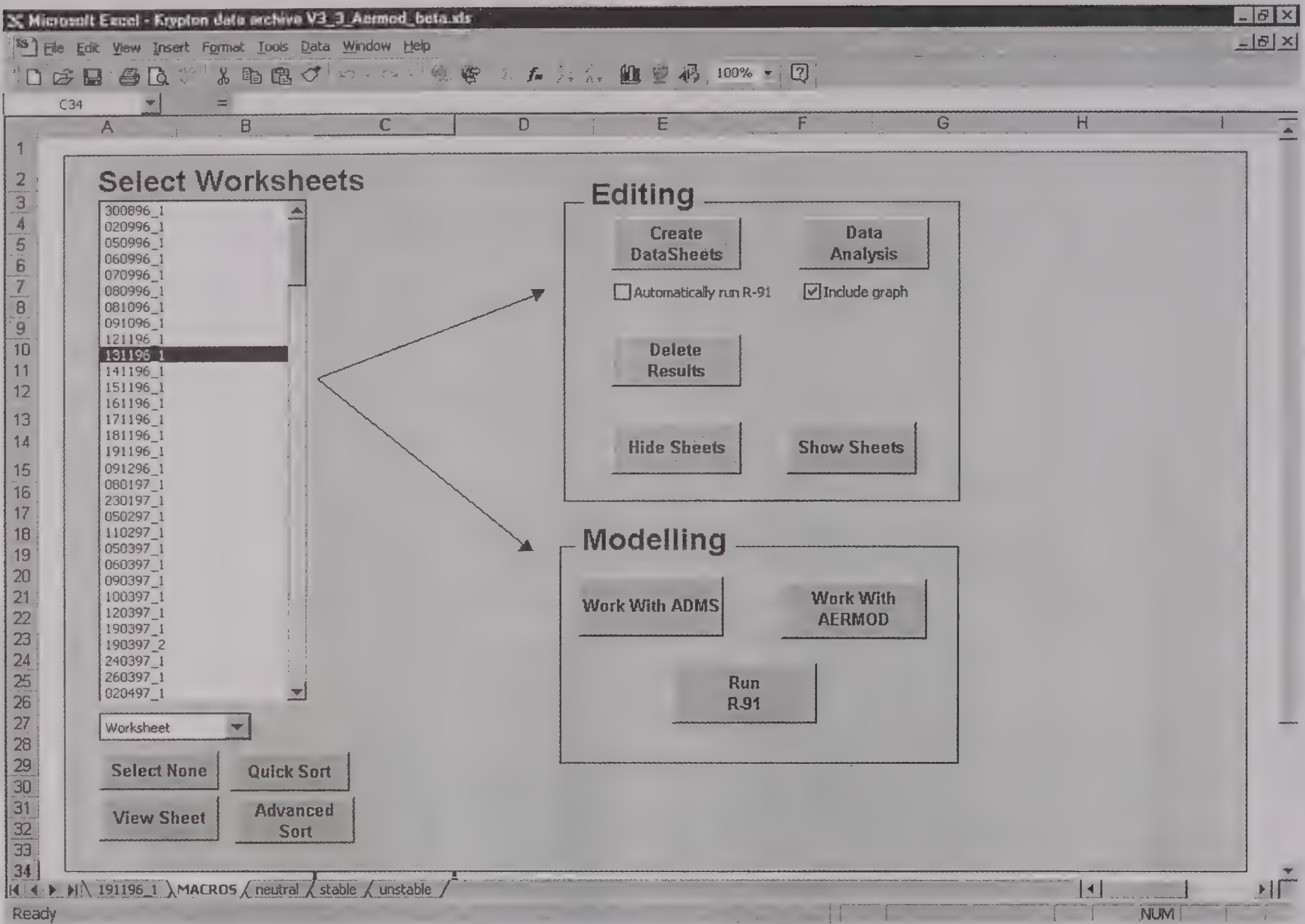

Figure 2: Screenshot of the model validation database used to evaluate the different model configurations.

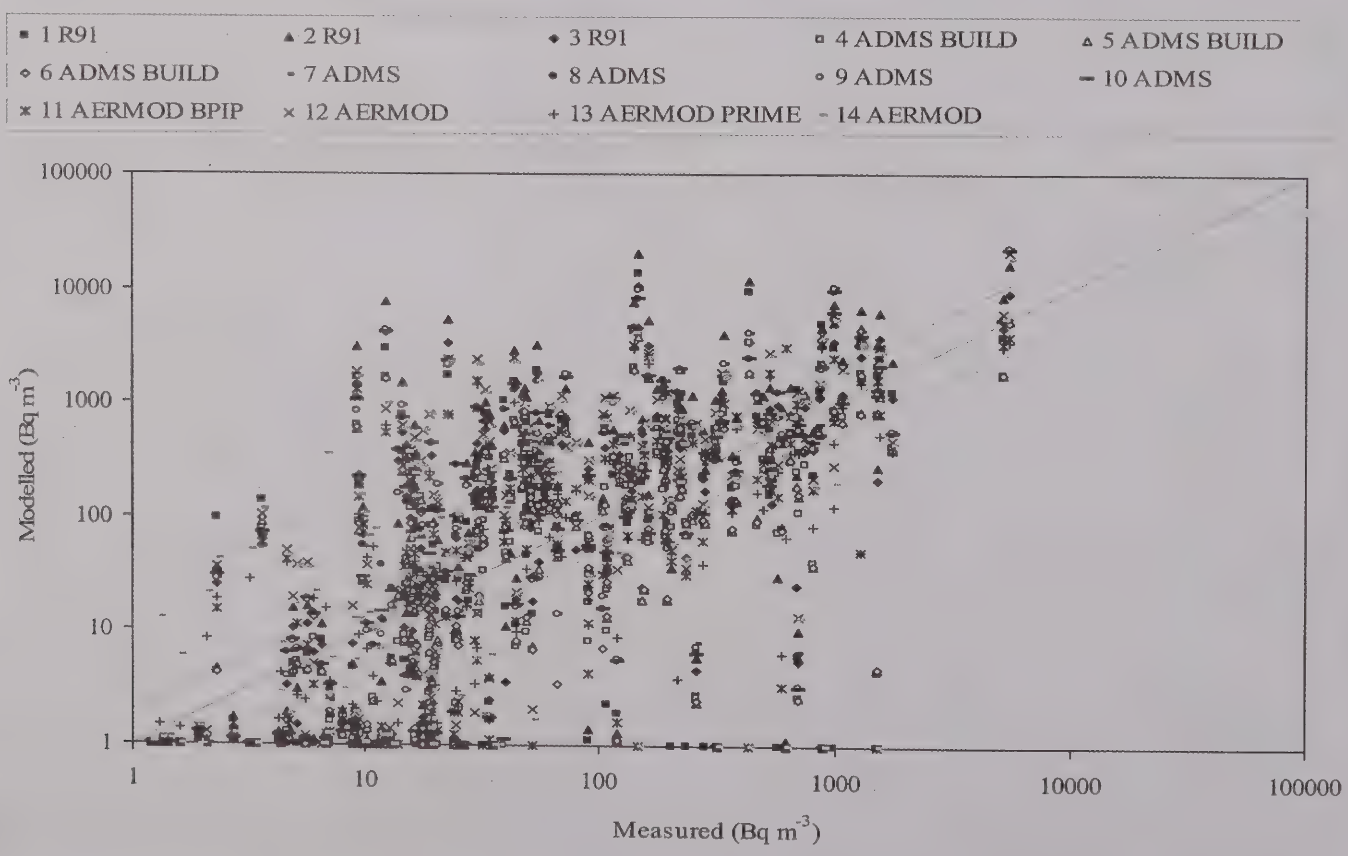

Figure 3: Scatterplot comparing the modelled and measured ^{85}Kr air concentrations (numbered as listed in Table 2).

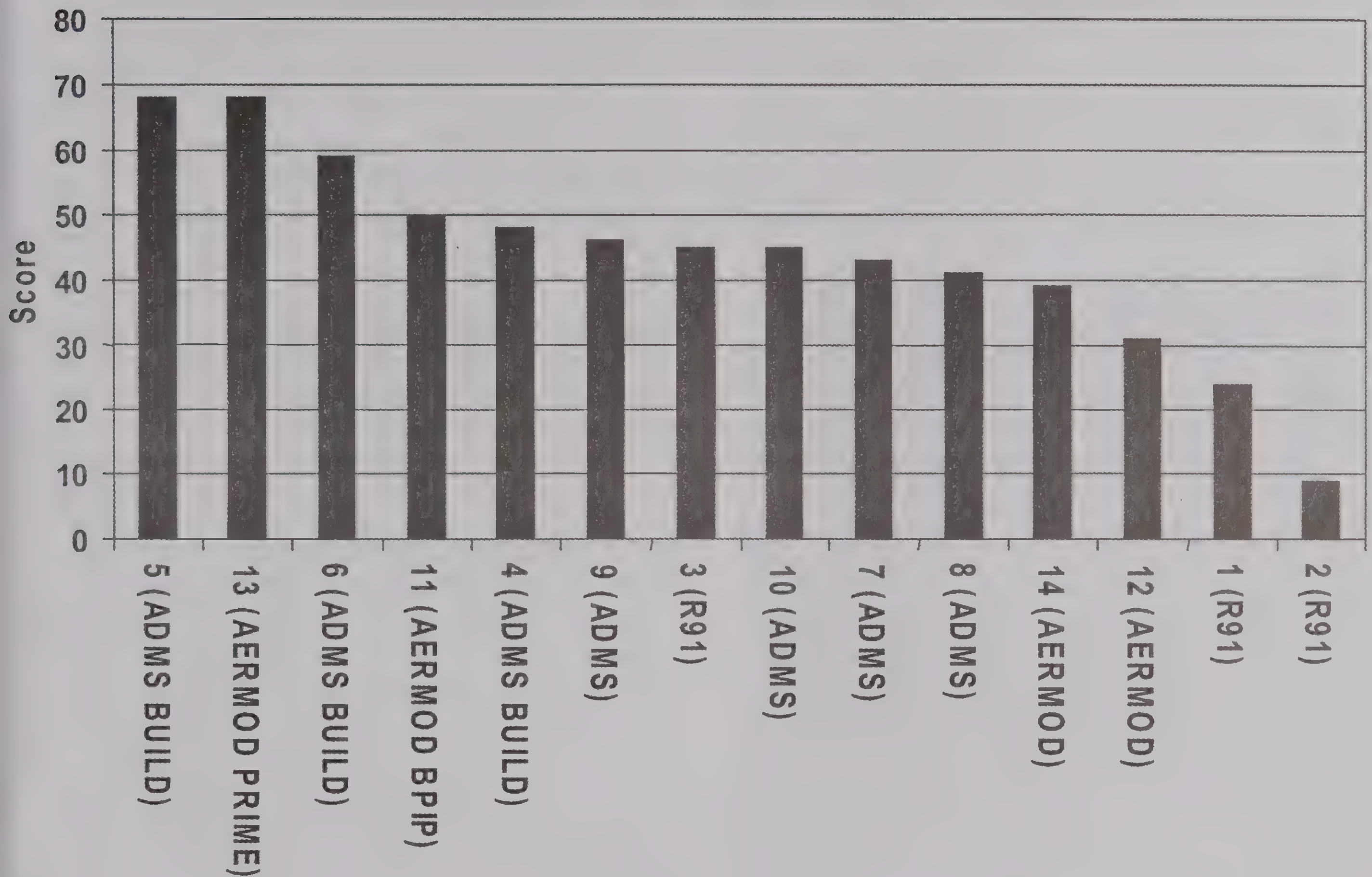

Figure 4: Ranked scores for the performance of each of the different model configurations (numbered as listed in Table 2).

Impact of Meteorological Dataset Choice on Dispersion Model Predictions

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As the modelling community is encouraged by regulatory bodies to consider uncertainty in their risk assessments, the use of representative meteorological data in atmospheric dispersion models becomes paramount. To combat the paucity of observational sites in certain areas of the UK, the Met Office now offer an alternative product, the NWP dataset, which comprises parameter values extracted from their mesoscale numerical weather prediction (NWP) model.

This paper compares site meteorological measurements from an emission site in West Cumbria, with predicted NWP parameters for the nearest representative NWP grid point, and with a Met Office weather dataset from Ringway, Manchester, one of the nearest meteorological stations, providing hourly sequential data for the equivalent period. It considers the influence of the choice of weather dataset on short and long term predictions from the ADMS dispersion model, specifically for an area of complex topography in a coastal region.

Introduction

Dispersion models are useful tools to help predict the environmental impact of atmospheric emissions from industrial operations. The dispersion of emitted species is critically dependent upon the meteorological conditions at the time of release, which are most commonly entered into commercial models as an annual hourly sequential data file.

Few industrial sites routinely monitor and record local site climatology in sufficient detail to collate a suitable meteorological dataset for input to atmospheric dispersion models. Where measurements are not available from the discharge site, model users typically purchase 'representative' records from suppliers such as the Met Office. Cloud cover observations and automated measurements of wind speed, direction, temperature and precipitation have been routinely recorded at Met Office weather stations around the UK for a number of years. Commercial dispersion models such as ADMS (CERC, 2001, Carruthers et al, 1994) have been designed with this in mind and contain algorithms using time of day and year in conjunction with cloud cover to estimate more continuous stability parameters such as boundary layer height and Monin Obukhov length.

As the Met Office upgrade monitoring equipment at UK stations, automated systems are becoming more prevalent and the use of observers less routine. The number of sites recording suitable cloud cover measurements (observations, cloud base level recorders or all sky cameras) is in decline, leaving large areas of the UK, particularly Scotland, Wales and northern England with few sites continuing to provide suitable data for dispersion modelling purposes. The 'ageing' of historic datasets that contain cloud cover observations has been previously addressed (ADMLC, 2002) and is not considered in this paper. The Met Office now offer numerical weather predictions (NWP) as an alternative to remote site data. The NWP dataset comprises parameter values extracted from their mesoscale Unified Model, which are subsequently processed to obtain model input parameters in a format suitable for use by the dispersion modelling community.

The Environment Agency's Air Quality Modelling Assessment Unit (AQMAU) encourages the submission of dispersion modelling assessments that include a discussion of the certainty associated with the model's deterministic predictions (EA, 2002). This requires the modeller to justify their choice of meteorological data as well as their treatment of other influential aspects such as buildings and terrain. A key question to consider is how representative is NWP data compared to site meteorological observations and remote site observations, specifically for use in dispersion modelling assessments. Some comparisons have been made between NWP datasets and Met Office observation records (Weaver, 2002) and agreement has been close for several locations. This paper includes a comparison of NWP data for 2 adjacent grids, Met Office observation records and on-site data, typical of that which may be obtained from an industrial site with its own automatic weather station. The site is situated in an area of complex topography, and is at least 150 km removed from the nearest Met Office observing station continuing to provide data for ADMS input.

The study compares meteorological parameters from on-site measurements at BNFL Sellafield, remote site data from Ringway, Manchester and NWP data for two grid points close to the Sellafield location and follows on to assess their influence on ADMS model predictions. The study is conducted in two parts, firstly a comparison of measurements of basic meteorological parameters (wind speed and direction, temperature and rainfall) and estimated stability parameters calculated within the ADMS pre-processor or by the flux profile method (Monteith & Unsworth, 1990) for site measured data. The second section compares the predictions of ADMS for a standard model scenario using each of the different meteorological inputs.

ADMS Meteorological Requirements

Dispersion models have traditionally used measurements of wind speed and direction (and rainfall for deposition

modelling) as input to dispersion algorithms. Further atmospheric parameters are required to provide information about the level of turbulence, the balance of heat transfer at the ground surface and the depth of the mixing layer. The use of simple Pasquill Gifford stability categories (Gifford, 1976) was common in earlier models, however new generation models such as ADMS accept a range of stability related parameters directly or, more typically, use cloud cover estimates to derive stability parameters within their own meteorological pre-processors. These stability parameters generally aim to describe the level of atmospheric turbulence through the magnitude of buoyancy and shear forces (Monin-Obukhov length, L_{MO}), the balance of heat transfer at the surface (sensible heat flux, $F_{\theta o}$), and the depth of the boundary layer (H) in which the pollutants are dispersed.

1.1 On-site Automated Weather Station Datasets

Meteorological measurements are routinely recorded by BNFL from their 50 m instrumented mast situated close to the Sellafield site (NGR 302140, 504630). Sellafield is a large and complex industrial area, situated on the (NW-SE) Cumbria coastline, less than 8 km to the west of the rising fells of the Lake District. It lies over 50 km from the nearest Met Office site providing historic data, and over 150 km from the nearest site continuing to provide data suitable for ADMS input. The mast is less than 1 km from the coastline, and obtains precipitation rate, solar radiation (K^+) and profiles of wind speed, direction and temperature measurements. ADMS datasets are created using either directly measured solar radiation input, or obtaining the Monin-Obukhov length and other parameters externally to ADMS using the flux profile method. Data from 1999 and 2000 was prepared for use in this study.

Despite careful siting of on-site automatic weather systems, the location is often operationally constrained by site boundaries and will inevitably be influenced by local structures and land use, the degree of influence differing with wind direction. Automated systems require routine servicing and calibration of instruments, and careful data processing and archiving to maintain quality and usability within dispersion models and for other purposes such as emergency planning. Measurements can be made at locations of particular interest, however, they are limited to providing information for one location in space, and uncertainty is introduced when extrapolating these data to provide a site climatology.

1.2 Meteorological Office Weather Station Datasets

Ringway data from 2000 was used within this study, being one of the closest operating sites to Sellafield providing hourly sequential data, although arguably not the most representative. The meteorological station is situated at Manchester Airport, with 30% of the area within 100m diameter of the site being occupied by buildings (Smith, 2002). The dataset contained low resolution (~±0.5 m/s) wind speed and high resolution (±0.1°C) temperature data. The ADMS meteorological pre-processor was used to estimate stability parameters from cloud cover input.

Met Office automatic weather stations and observing sites are likely to be located in areas where local small scale influences are minimised and hence provide an accurate picture of boundary layer conditions for the area of measurement. However, the datasets are inherently unable to fully reproduce the climatology of the industrial site under consideration, due to dissimilarities of local topography and land use and the variation of synoptic scale meteorological systems as they pass between the sites. These factors will inevitably introduce uncertainty into any model predictions.

1.3 Meteorological Office NWP Datasets

The mesoscale version of the 'Unified Model' is an NWP tool developed and used by the Met Office to help provide detailed forecasts for the UK. A wide range of observed and automated data is fed into this model to help refine forecasts that are driven by the solution of three dimensional atmospheric dynamic flow equations. Although designed for forecasting purposes, NWP data has been used extensively for other applications, and is now being promoted for use in dispersion models.

The data purchaser chooses the most appropriate grid point to represent their site, NWP data is then extracted and post processed to provide a dataset of identical format to those from observing and automated stations. It is understood that this involves converting cloud cover information from three heights into a total cloud cover parameter, and extrapolating three hourly data into hourly values. It is expected that the post processing could be minimised in future with increasing computer power and the provision of alternative parameters directly available from NWP output, such as mixing depth, friction velocity and sensible heat flux. If sufficient information is extracted from the NWP model, it may reduce the need for dispersion models to complete any pre-processing of NWP data.

NWP model output parameters reflect grid averaged conditions, the current grid size being 11x11 km^2, with a vertical resolution of 38 levels. At its present resolution, the model cannot resolve local topographical features at a fine level, and the predicted wind field is not able to describe the exact conditions in areas of complex terrain. However, it may still be preferable to using data from a site that is known to be remote and less than ideal in terms of representivity of industrial site conditions. Improvements to the NWP model are likely to include a higher resolution of 4x4 km in the next few years. Advantages of the NWP dataset include the continuity of data, with very few data gaps and removal of problems associated with instrument thresholds and deterioration in performance.

Data from two consecutive grids were used in this study to investigate sensitivity to grid choice. NWP grid point A was located 5.8 km from Sellafield, representing an area of similarly flat open terrain close to the coast, grid point B was situated 10 km from site, close to an escarpment running west to east inland from the coastline.

Comparison of Meteorological Datasets

Wind roses for each of the datasets are displayed in Figure 1. All datasets had over 99% coverage, with a maximum of 5% 'calm' conditions, specified as 'calm' when speeds were less than 0.75 m/s, i.e. below the critical ADMS threshold. These wind roses were produced using the ADMS wind rose function and, although the roses for Sellafield site measurements indicate obvious differences in the wind field between 1999 and 2000, a number of consistencies are observed. Both years demonstrate a high variability with a predominance towards flow from the south east, south west and west north west. Northerly and easterly winds are light (less than 5 m/s) and infrequent, while westerly winds are invariably strong. The NWP datasets exhibit a similarly infrequent north and easterly flow to that observed in site measurements, while Ringway data displays a strong dominant southerly flow. Met Office datasets do not record high speed westerly winds from the 270°-300° sector as frequently as those observed at Sellafield. These conditions are most likely related to sea breeze coastal effects throughout summer which are unlikely to occur at Ringway and are not clearly resolved by the NWP model.

Differences between the wind fields from NWP grid points A & B were found to be larger than the inter-annual difference observed between 1999 and 2000 at each grid. Predictions for the NWP B grid point display a tendency towards lighter winds (< 5 m/s) and more frequent east south easterly flow than all other datasets. Overall, NWP grid point A appears to match the site measured wind field more closely.

Figure 2 shows the annual frequency of wind speeds falling in binned speed categories. The NWP datasets reflect similar conditions to those recorded at Sellafield, with the exception being the less frequent high wind speed conditions (<8.2 m/s) and lower maximum wind speeds. NWP data provides an average value over the grid area, hence peak values are likely to be lower than those observed in measured datasets. Ringway data also displays a lower frequency of the highest wind category compared to Sellafield – this may be an artefact of the dataset and its low resolution wind speeds.

Statistical analysis revealed strong agreement between all datasets throughout the year with the largest mean bias between NWP B and the Sellafield measurements of 17% (NWP predicting lower values than measured). The lowest correlation was between Ringway and Sellafield measurements (correlation coefficient $R^2 = 0.61$), while the NWP B data showed the lowest percentage (38%) of values within a factor of 1.25 of the Sellafield measurements. Overall, the NWP A grid point displayed strongest agreement with Sellafield measurements.

Agreement of temperature values between all datasets was generally very close, with mean bias between datasets always under 5%, correlation coefficients of between 0.90 and 0.95, and over 70% of time matched values from all datasets within a factor of 1.25. Both NWP grid points displayed similar inter-annual trends to those in the Sellafield site. Agreement of total annual rainfall was within 25%, reflecting the high spatial and temporal variability of precipitation. Similar grid averaging effects were observed with temperature and rainfall predictions, with NWP data providing less extreme events than those recorded in the Sellafield measurements.

Stability parameters are not routinely measured and observations of cloud cover or measurements of solar radiation are commonly input to the ADMS meteorological pre-processor which then estimates parameters such as sensible heat flux, Monin Obukhov length and boundary layer depth. Pre-processor outputs from site measurements (K^+ or L_{MO} input) and NWP and off-site measurements (cloud cover input) were compared. Agreement between NWP dataset parameters was very close, compared to the differences between the Sellafield L_{MO} and Sellafield K^+ processed parameter output, suggesting that the choice of input parameters (and hence reliance on associated algorithms) might have a stronger influence on stability parameters than the source of the original data. The use of the K^+ input resulted in a significantly higher frequency of stable conditions than that estimated from NWP data, while the use of L_{MO} input resulted in increased occurrence of unstable conditions.

Comparison of ADMS Predictions

The study was limited to one standard model scenario, using a buoyant release (130 C exit temperature and 25 m/s velocity) of NO_2 at 1 g/s from a 40 m stack, a roughness length of 0.3 m and a variable grid resolution (between 50 and 1000 m). All other model parameters remained as default values.

Table 1 provides the predicted maximum annual average, hourly and percentile concentrations from ADMS runs using each of the year 2000 datasets. Similar trends are observed in both the short and long term averages. The highest long and short term concentrations are predicted for the Sellafield L_{MO} dataset where the L_{MO} was calculated using the flux profile method prior to inputting met data to ADMS. Annual average predictions from all datasets lie within a factor of two and short term predictions within a factor of three. Consistently, the top 0.2% of hours from each dataset occurred during convective periods, mostly around mid-day, with light winds generally under 1 m/s, clear skies and warm temperatures for the season. The agreement between the NWP sites is very strong as expected due to the grids being adjacent. Predictions of short term peak concentrations using NWP, Ringway or the Sellafield solar radiation dataset are in close agreement, with NWP predicting slightly higher values (less than a factor of 1.6) than the site measurements. The largest difference in short term percentile concentrations is observed between predictions from the site measurements using K^+ directly and those where the L_{MO} was calculated using the flux profile method prior to inputting met data to ADMS. This reflects the less frequent unstable conditions observed in the K^+ dataset, resulting in generally lower maximum percentiles.

Table 1: ADMS predicted maximum concentrations for year 2000 datasets (μg/m³)

	Max annual average	Max hourly	99.80% hourly	95% hourly	68% hourly
Sellafield L_{MO} 2000	0.54	25.25	14.08	5.84	1.3E-04
Sellafield K^{+} 2000	0.34	9.38	6.83	2.80	3.3E-07
NWP A 2000	0.39	13.43	10.67	3.10	3.7E-06
NWP B 2000	0.30	13.71	9.03	2.70	2.2E-09
Ringway 2000	0.46	11.85	7.71	3.20	2.4E-03

Contour plots of predicted maximum annual average concentrations from all year 2000 datasets are shown in Figure 3. The footprints of concentrations from model runs using both Sellafield met inputs are similar reflecting the identical wind field input. The NWP A model results also bear a strong similarity in overall coverage, reflecting the strongest similarity to wind fields recorded on site. The two main peaks to the north east and east south east are observed in all results, however, differences in wind fields for the NWP B and Ringway datasets are apparent in their dissimilar footprints. The downwind distance of peak concentrations is similar throughout all datasets with the exception of the L_{MO} dataset, where highest concentrations are observed closer to the source due to differences in stability conditions predicted using the flux profile method and the ADMS pre-processor algorithms.

Conclusions

Meteorological observations from an instrumented mast close to the BNFL Sellafield site on the West Cumbrian coast were compared with two years of NWP data from the most representative local NWP grid points and one year of Ringway data from one of the closest Met Office weather stations. Agreement between measured and predicted wind speed, direction, temperature and precipitation was generally good, with closer agreement between NWP grid data and on-site measurements than between any combination that included Ringway data. The choice of NWP grid point influenced the agreement with measured wind fields and resulting concentration contours, however, the impact on predicted maximum concentrations was small. Measured data displayed more frequent and extreme conditions such as high winds and heavy rainfall than those predicted by NWP, demonstrating differences between grid averaged data (NWP) and 'spot' measurements. The measured wind field also exhibited more frequent strong westerly winds than NWP, suggesting the model data may not fully reproduce wind fields that are influenced by local topography such as coastlines. The inability of the NWP data to resolve accurately topographically influenced wind fields may affect the location of and, to a lesser extent, the magnitude of the maximum annual average and its surrounding contour footprint.

Comparison of turbulence and boundary layer parameters derived from measured and NWP datasets revealed larger discrepancies than between directly measured parameters, highlighting the influence of the input parameter choice (e.g. cloud cover, solar radiation or profile measurements) and the corresponding algorithms used to estimate stability parameters. This has an observable impact on short term concentrations.

NWP data provides area average information, currently for an 11x11 km² grid. For commercial dispersion models such as ADMS and AERMOD, a model domain of a comparable or smaller size is typically chosen to inform the modeller of the effects of industrial site emissions at local sensitive receptors. An advantage of NWP area average met data is that parameters are likely to be appropriate (on average) throughout the plume's travel, allowing the dispersion model to then treat the impact of very local influences such as buildings and land use separately. Measured datasets provide information on the state of the atmosphere at one point in space, which is then assumed to be typical of the local atmosphere in the vicinity of the release point and plume location. In this case, it may be argued that it is less able to provide an appropriate average value than the NWP data.

This study has assessed the performance of NWP extracted data within dispersion models. The site chosen for comparison was driven by the availability of independent on-site met data, and by the fact that the site was located at a significant distance from any current Met Office weather station site providing full ADMS parameters. In addition, the site is situated in complex topography, lying between the Cumbrian coastline and Lake District fells. It can be argued that if the performance of the NWP data for this complicated location is considered fit for purpose, then it follows that its use at less complex sites, situated close to Met Office weather stations, is likely to be more easily justified. The differences between ADMS dispersion predictions using on-site, remote site or NWP data were found to be within the level of uncertainty normally associated with dispersion model results.

A site specific assessment should be made when choosing appropriate met data, to consider the quality of data obtained on site, the representivity of available Met Office automated and observed data and the limitations of using NWP data for the site in question. The choice of meteorological dataset in modelling assessments is often dictated by practical constraints, i.e. availability, quality, delivery and cost. The suitability of the data may necessarily take a lower priority, and it should be remembered that choice and use of met data is just one of a number of uncertainties in a modelling assessment, with other major contributions being the emission rates, treatment of complex effects such as buildings, empirical algorithms within the pre-processor and dispersion codes, and model domain choice.

Availability of NWP data was limited when this study began; however, as communication between the dispersion modelling community and the Met Office continues, and as the NWP grid resolution increases, it is hoped and anticipated that improvements to the provision of NWP data, including optimisation of provided parameters, will be achieved, and the use of NWP data will be more widely accepted.

This article has used findings from an MSc research project (Kidd, 2002) conducted at Westlakes Research Institute in collaboration with the University of Reading. Data was kindly provided by BNFL, Sellafield and the Met Office.

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Figure 1: Wind roses of Sellafield, NWP and Ringway datasets

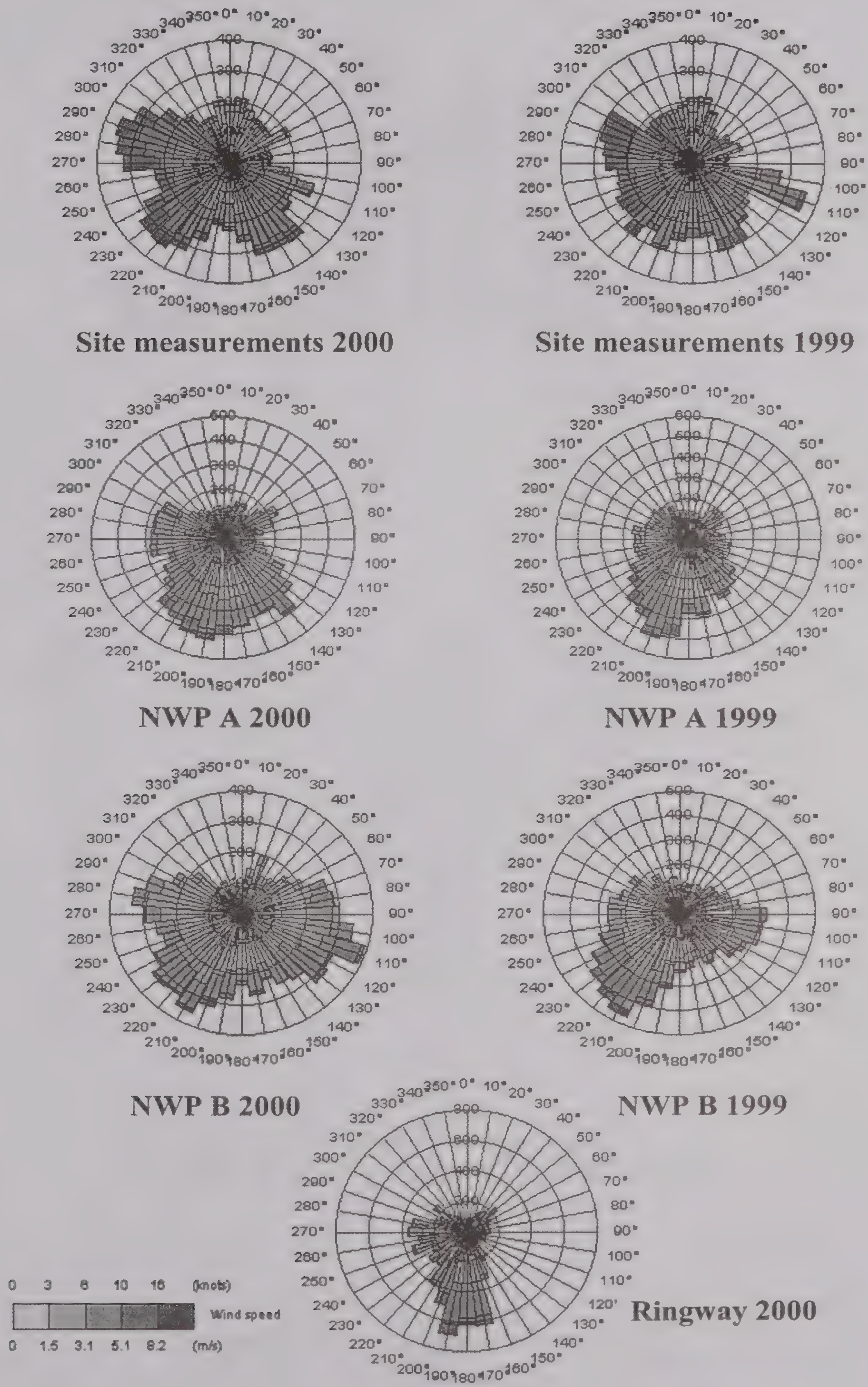


Figure 2: Frequency of binned wind speed categories for each dataset

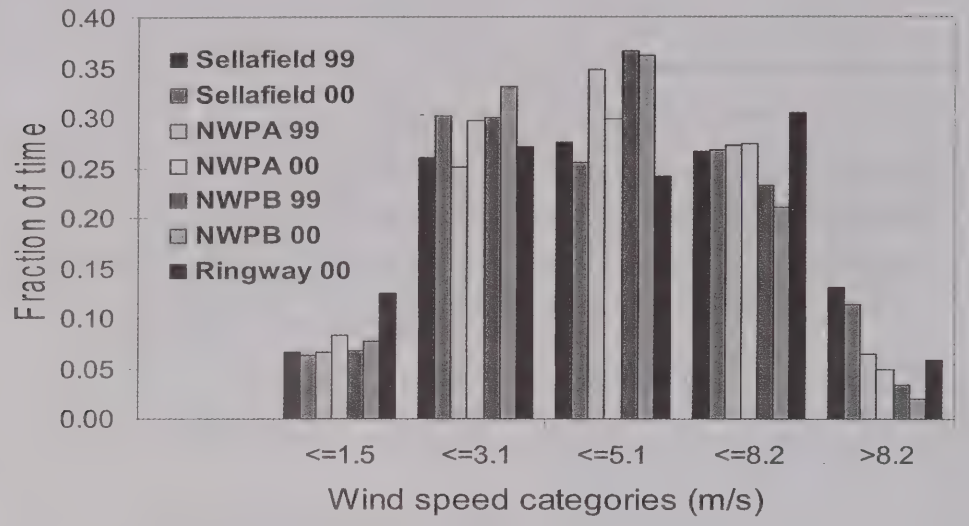
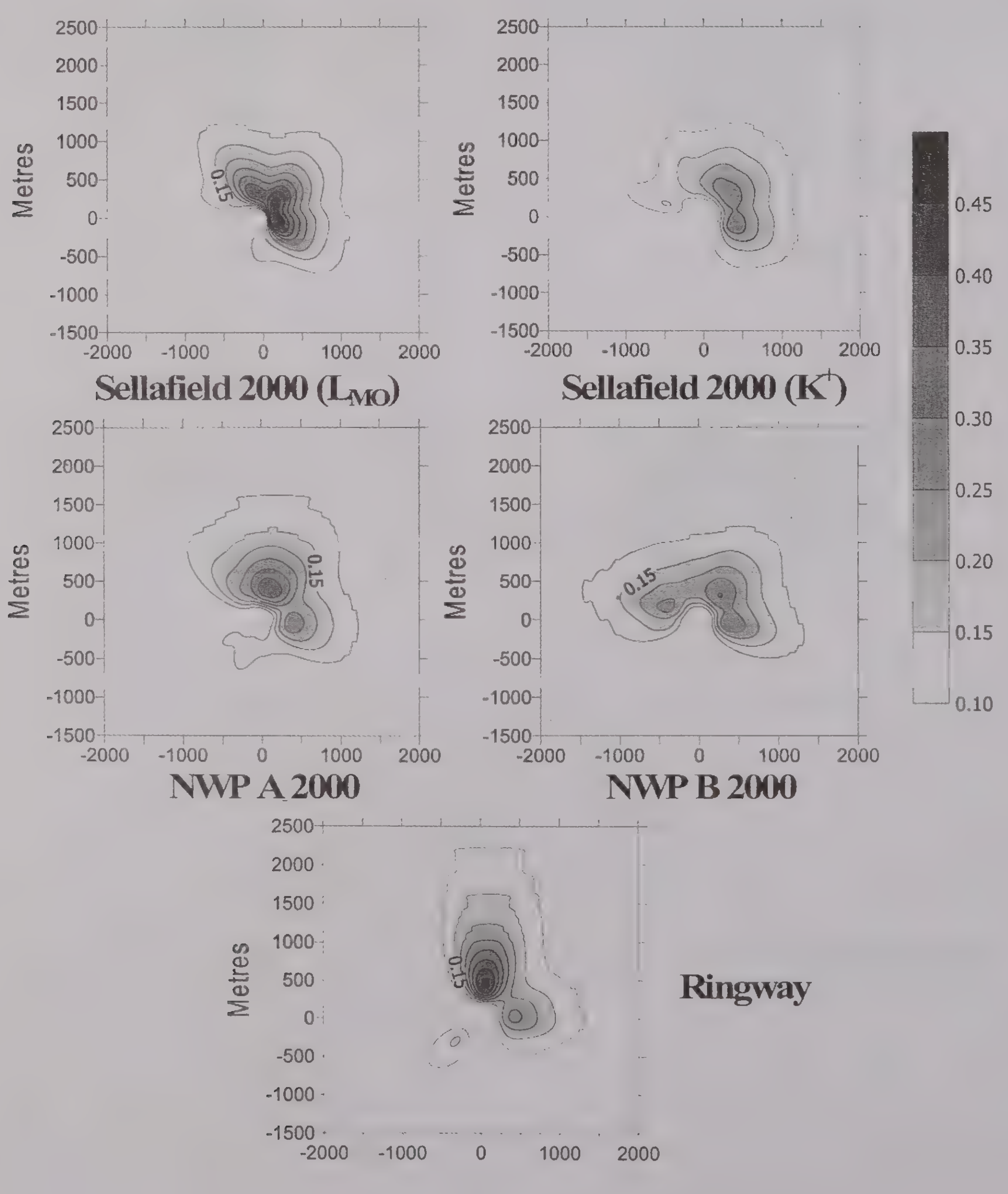


Figure 3: Annual average concentration contour plots for year 2000 datasets



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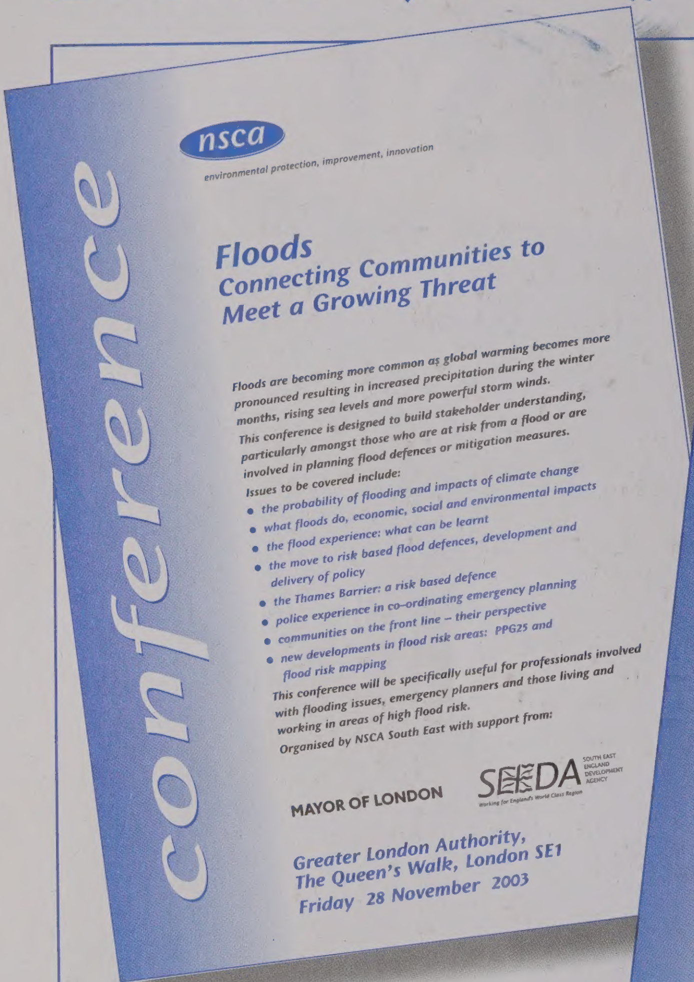
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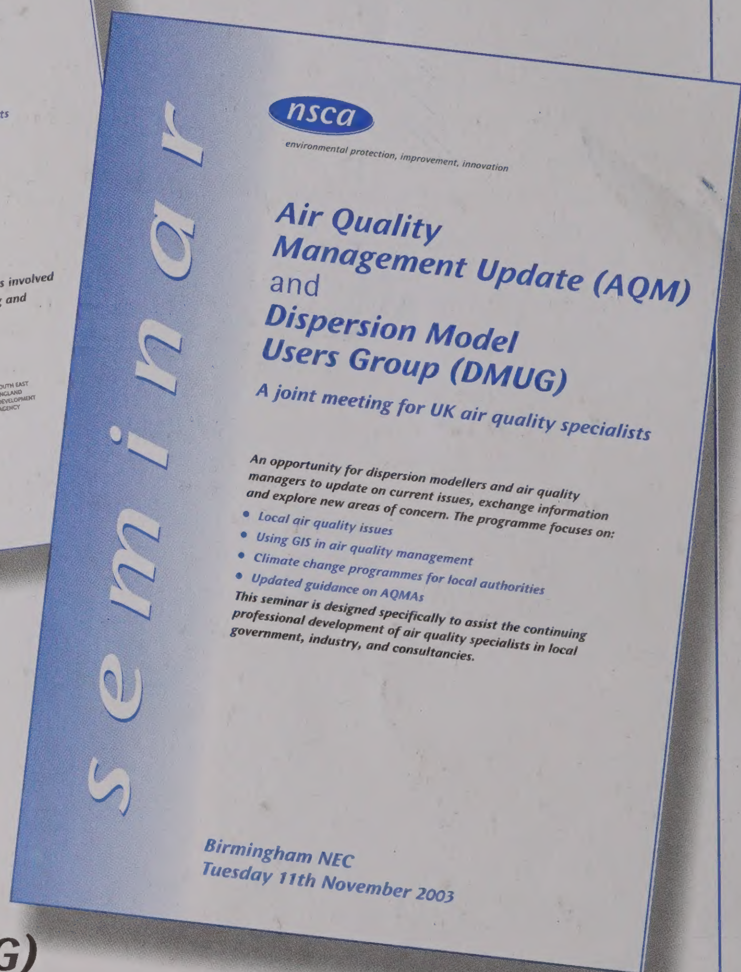
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